Implications of climate change impacts on fisheries resources of northern Australia

Part 1: Vulnerability assessment and adaptation options

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Project No. 2010/565

The Fisheries Research and Development Corporation plans, invests in and manages fisheries research and development throughout Australia. It is a statutory authority within the portfolio of the federal Minister for Agriculture, Fisheries and Forestry, jointly funded by the Australian Government and the fishing industry.

Cover photos: Torres Strait fishing boat with dory (J. Johnson); Recreational fishers with red emperor (Tippo); Seagrass meadows (GBRMPA).

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1 NON TECHNICAL SUMMARY

2010/565	Implications of climate change impacts on fisheries resources of northern Australia.

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OBJECTIVES:
1. Describe the projected climate-driven changes that are relevant to northern Australian fisheries resources.
2. Assess the potential impacts of climate change on key fisheries and species in northern Australia.
3. Identify approaches that are adaptive to potential climate change scenarios.

OUTCOMES ACHIEVED TO DATE

Provision of scenario-driven recommendations of adaptive management approaches that provide for the sustainability of northern Australia fisheries in a changing climate.

- The final project workshops worked with stakeholders to identify adaptation options based on likely future fishery scenarios. Scenarios were based on the reviews of species biology and ecology, as well as future localised climate projections, and described the likely response of key species to climate change. For example, the abundance of barramundi on the east coast is likely to decrease by 2030 due to reduced rainfall and increased water extraction, as well as habitat changes. Adaptation options across all species were grouped as: Alteration of fishing operations, Management-based options, Research and Development and Looking for Alternatives. These groupings generalise the types of adaptation that fishers and managers identified and species-specific options are also given in the report appendices. With these options stakeholders also identified the likely barriers and who is responsible for their implementation. Cost was identified as a key barrier to most options as well as political opposition. The options presented here represent an initial, but important, step towards northern Australian fisheries preparing for climate change.

Determination of the vulnerability of northern Australia's fisheries to climate change.

- A key output from the project was the development and application of vulnerability assessments of key fishery species from three key regions of northern Australia. The assessment framework developed is semi-quantitative and draws on the elements of exposure, sensitivity and adaptive capacity. The assessments are species-based and regionally targeted and he framework is a tool to assess the relative vulnerability of species to climate change, providing an objective and strategic basis for developing responses to projected changes. The framework is also transparent and provides the means for determining the appropriateness of responses. The framework can readily be adopted for similar assessments in other regions and, with modification, could also be adopted in other disciplines. The vulnerability assessments here focused on 2030, a medium-term outlook, and one considered to be more relevant to all stakeholders, although an assessment was also carried out based on the A1FI emissions scenario for 2070.

Greater understanding of the impacts of short and long term climate variability on northern Australia's key fisheries species, fisheries and regions of northern Australia, and the key environmental drivers. These include identification of priority species, fisheries and/or locations for targeted monitoring.

- The project has delivered as a major output, summary tables of the likely impacts of climate change on key northern Australian fishery species and habitats, also identifying the environmental variables of significance. This was done for three regional areas of northern Australia based on projected climate change for 2030. The key species likely to be impacted by changes predicted for 2070 (A1FI emissions scenario) were also identified. The vulnerability assessment process also prioritised species for action.

Generally, inshore species were assessed to be more likely to be affected by future climate change. The east coast was identified as a critical region given that rainfall (riverflow) is projected to decrease and many species populations are known to be positively associated with riverflow. This is amplified by the likely increase in water extraction for land-based uses, particularly on the east coast. Across all regions in northern Australia the species identified as highest priority (high vulnerability and high fishery importance) were: golden snapper, king threadfin, sandfish, black teatfish, tiger prawn, banana prawn, barramundi and mangrove jack.

Improved capacity for fisheries management agencies and industry to assess current practices and policies to optimise positioning for future predicted scenarios.

- Collectively, the key outputs of this project provide an informed basis for management and industry to assess current fisheries management against likely future scenarios. Management as well as commercial and recreational fishing interests were key participants in the project and had direct input into key outcomes providing a credible base for further extension and uptake by relevant fishery stakeholders.

NON TECHNICAL SUMMARY:

Climate change is a major environmental threat and there is a national imperative to determine likely impacts on fisheries in Australia. Northern Australia is predicted to be affected by increased water temperatures, changes in rainfall patterns and resultant increases in river flows to the marine environment, increased intensity of cyclones, ocean acidification, and altered current patterns, which will also affect habitats. These changes will directly and indirectly impact on fishery species including modified phenology and physiology, altered ranges and distributions, composition and interactions within communities, and fisheries catch rates.

For fishery sectors in northern Australia to be able to respond positively and adapt to climate-induced changes on fish stocks there is a need to determine which stocks, and where, when and how they are likely to be affected, and prioritise species for further actions. This project set out to do this using a structured approach to develop and carry out a semi-quantitative vulnerability assessment and conduct stakeholder workshops to identify adaptation options. These outputs were informed by several tasks: a descriptions of past and future climate; identify likely impacts of climate change on habitats and key fishery species; detailed species profiles to document and understand key fisheries, species life histories, and sensitivity to environmental variability (see Part 2 companion report); and analyses of existing data sets of key species to better understand sensitivity to environmental change.

Changes in climate across northern Australia are predicted to be highly variable depending on the specific region with the trend for warmer, less saline and more acidic waters, rising

sea levels, more intense cyclones and changed oceanographic conditions. By 2030, north-western Australian sea surface temperature (SST) will be 0.6 – 0.9 °C warmer, the Gulf of Carpentaria will be 0.3 – 0.6 °C warmer, and both regions will have similar or slightly higher rainfall (0 – 5%)(and riverflow). Sea level is projected to rise between 10 and 20 cm and there will be a weakening of the Leeuwin current on the west coast. By 2030, east coast SST will be 0.3 – 0.6 °C warmer, there will be -10 – 0 % less rainfall (and riverflow), sea level will rise between 5 and 15 cm and the East Australian Current with strengthen. Fishery species will be directly exposed to these changes and will also be indirectly exposed to impacts on habitats.

A literature review of climate effects on habitats in northern Australia examined the key habitat types: coral reefs, seagrass meadows, mangroves, floodplains, coastal bays and estuaries. The review found that projected increases in SST will cause more coral bleaching, and ocean acidification will reduce coral growth and structural integrity, resulting in a loss of reef diversity and structure. Increased storm severity and extreme riverflow events, resulting in increased turbidity and reduced solar radiation, will reduce seagrass cover and species diversity. Sea-level rise may result in a landward migration of mangroves depending on localised barriers and, coupled with altered rainfall patterns, will change the connectivity between rivers and floodplains, resulting in the potential loss of freshwater floodplains.

To prioritise species to be potentially included in the project we consulted stakeholders and, based on a combination of fishery importance and perceived sensitivity to environmental variation, identified a total of 47 key fishery species across the three regions of northern Australia – east coast (40 species), Gulf of Carpentaria (36 species) and north-western Australia (37 species). The species that were generally ranked highest across the three regions included barramundi, mud crab, banana and tiger prawns, coral trout, golden snapper, black jewfish, Spanish mackerel and king threadfin.

Analyses of existing data sets were carried out on barramundi, red throat emperor, coral trout, saucer scallop, Spanish mackerel, and golden snapper to identify correlations between recruitment and/or catch rates with particular environmental variables. A positive correlation was found between barramundi CPUE and river height as well as rainfall in the Northern Territory, providing further evidence of the positive influence of rainfall, riverflow (and floodplain inundation) on barramundi catchability and possibly recruitment. In southeast Queensland saucer scallop recruitment was enhanced in years of cooler water. Recruitment also appeared to be positively influenced by higher local riverflow and by the presence of a cyclonic current eddy in the Capricorn region. Recruitment of Spanish mackerel on the Queensland east coast appeared to be linked to SST with cooler years positively influencing recruitment, although the causal mechanism for this relationship is unclear. Analyses of Spanish mackerel data supported the hypothesis of a single east coast stock. Analyses of the other species produced equivocal results.

Region-based ecological vulnerability assessments for climate change scenarios for 2030 were carried out for the three regions (i) north-western Australia (23 species), (ii) the Gulf of Carpentaria (21 species), and (iii) the Queensland east coast (24 species). Species with the highest ecological vulnerability to climate change tended to have one or more of the following attributes: an estuarine/nearshore habitat preference during part of their life cycle; poor mobility; reliance on habitat types predicted to be most impacted by climate change; low productivity (i.e., slow growth/late maturing/low fecundity); known to be affected by environmental drivers; and fully or overfished. Based on the combination of ecological vulnerability to climate change and fishery importance, the highest priority species were identified as: golden snapper, king threadfin, sandfish, black teatfish, tiger prawn, banana prawn, barramundi, white teatfish and mangrove jack.

In the medium-term (2030), the most common impact identified across all species was reduced size of populations due mainly to lower rainfall and riverflow, which affects primary productivity and therefore survival of early life history stages. The indirect effects of habitat degradation on key life history stages and increasing SST were also likely to impact some species by 2030. In the longer-term (2070), changes in rainfall/riverflow, SST and habitats will continue to impact species, with ocean acidification and salinity likely to increasingly become factors that impact species through disruption of early life history development (e.g. coral trout) and habitat effects (particularly coral reefs). Some species in some regions, for example banana prawns in the Gulf of Carpentaria, may experience higher population sizes due to projected increases in rainfall. Individual species and the likely impacts on them as a consequence of changed climate are discussed in detail in the report.

Rainfall and riverflow are key environmental drivers for many fisheries populations in northern Australia through enhancement of local primary productivity and larval/juvenile survival, and by connecting key habitats such as estuaries and floodplains. The Queensland east coast in particular is a key area for concern due to projected lower rainfall and more extreme (i.e., longer) wet and dry periods, coupled with the expected increase in water extraction for land-based use. Many fishery species of northern Australia use estuarine, floodplain and nearshore habitats and so are likely to be impacted by changed hydrological conditions, particularly barramundi that use all habitats during various stages of their life history. For example, longer periods of wet and dry will result in higher variability in the size of barramundi populations. The project found there was a high level of uncertainty in how individual species, particularly their early life history stages, will be affected by changed SST, pH and salinity.

Based on priority species for each region and the likely impacts on these species, we presented future scenarios to stakeholders at workshops conducted in Darwin and Townsville. Through discussion these stakeholders identified a range of potential adaptation

options. We were able to group the options identified into four categories: (1) Alteration of fishing operations, (2) Management-based options, (3) Research and Development, and (4) Looking for alternatives. Examples of the types of options stakeholders identified include: modification of target species and/or gears, revised size/catch limits, habitat protection, targeted monitoring of species, codes of conduct, restocking and habitat restoration. Most of the adaptation options identified involved regulatory changes and/or policy decision-making (Management-based options). Stakeholders also identified that major barriers to adaptation for northern Australian fisheries were likely to be costs, political opposition and bureaucracy. In terms of responsibility for taking actions, it was acknowledged that all stakeholders will need to play a role, however government will need to need to be a lead player in this process.

Due to the number of fishery species assessed across a vast area, this project took a broad approach to determining the relative vulnerability of key fishery species in northern Australia. Despite this, the project developed a process to prioritise these species to identify likely impacts on key species based on the best available knowledge, and to also engage stakeholders in identifying the range of potential adaptation responses that would mitigate consequences both environmentally and on the fisheries and its participants. To further develop adaptation options we suggest the need for a regional focus with strong representation of all relevant stakeholder groups and multiple workshops that consider: priority species and likely impacts identified in this project (as well as the underlying mechanisms behind the impacts), and current management and government policy. There is also a need to rigorously prioritise adaptation options, identify complementarity among regions and species, and to identify clear pathways for adoption. Building a solid business case for each option that articulates costs and tangible benefits will maximise the likelihood of the commitment of the associated resources required for successful adoption.

2 ACKNOWLEDGEMENTS

3 STRUCTURE OF THIS REPORT

There are two parts of this report. Part 1: Vulnerability assessment and adaptation options, and Part 2: Species profiles. Part 1 (this report) represents the main body of the project reporting on the approach taken in carrying out the vulnerability assessments, the results and discussion of these results. The project was structured into multiple tasks that lead to the identification of the types of adaptation options that northern Australian fisheries may need to adopt in the future under current climate change projections and potential impacts on species (see Figure 7.2). Part 1 outlines the detail of each of these tasks except for the reviews of key northern Australian fisheries species, which are presented in a separate volume; Part 2. Part 2 describes in detail the fisheries, biology, ecology and life cycle, and sensitivity to environmental variability for 23 different species/species groups; 8 invertebrates and 15 finfish and sharks. These species profiles provide much of the information that supports the project vulnerability assessments, the identification of likely impacts on species, and adaptation options. Part 2 also represents a valuable stand-alone resource for any fishery stakeholder.

4 BACKGROUND

This application for this project was developed through consultation and in conjunction with industry (Queensland Seafood Industry Association, Sunfish, Amateur Fishermans Association of the Northern Territory, Northern Territory Seafood Council), oceanographic scientists and modelers (Craig Steinberg & Richard Brinkman, Australian Institute of Marine Science), research scientists with relevant experience (Julie Robins, QDAFF; Andrew Tobin, JCU; Thor Saunders, NT Department of Primary Industries and Fisheries; Stewart Frusher, Tasmanian Aquaculture and Fisheries Institute; Bill Sawynok, Recfishing Research/Capreef; Nick Caputi, WA Department of Fisheries), and resource managers from Queensland and the Northern Territory (Mark Lightowler, John Robertson & Warwick Nash, QDAFF; Randall Owens, Rachel Pears, GBRMPA; Julia Playford, DSITTA; Steven Matthews and Andria Handley, NT DPIF). Several of these key scientists and end-users were co-investigators on the project.

During this project there was ongoing consultation and collaboration with similar projects in South-eastern Australia (PI's Gretta Pecl and Tim Ward; FRDC SE climate change adaptation project) and Western Australia (PI Nick Caputi; FRDC WA climate change adaptation project) to facilitate ongoing learning that will optimise and standardise the approaches taken across all projects. We also maintained contact with concurrent research projects to use results as relevant: coral trout, Professor Morgan Pratchett; barramundi, Professor Dean Jerry. Also, the results from this project provided valuable input into the FRDC project assessing socio-economic impacts of climate change on fisheries across Australia (PI Stewart Frusher).

This project relates directly to several other completed projects. These include the recently completed AFMA project (2013/0014) 'Assessing the vulnerability of Torres Strait fisheries and supporting habitats to climate change' (Welch and Johnson, 2013), the Great Barrier Reef climate change vulnerability assessment (Johnson and Marshall, 2007) and the Pacific climate change vulnerability assessment of fisheries (Bell et al, 2011), and several FRDC projects including: (2008/103) 'Adapting to change: minimising uncertainty about the effects of rapidly-changing environmental conditions on the Queensland coral reef finfish fishery' (Tobin et al. 2010); (2001/022) 'Environmental flows for sub-tropical estuaries: understanding the freshwater needs of estuaries for sustainable fisheries production and assessing the impacts of water regulation' (Halliday and Robins 2007), and the recently completed QDEEDI/QDNR/QCCCE/JCU project that examined short- and long-term climate variability on barramundi fisheries. Each of these studies provided important case studies and templates for analytical approaches adopted during the current project, and provided a solid basis from which to extend understanding of the phenology of selected key species and to examine potential future impacts under climate change scenarios. The project drew on many data sets collected over many years from past research projects and on-going monitoring programs including fisheries-related and environmental-related data sets.

5 NEED

Climate variability has always been an influence on fisheries productivity however the current trends and rates of change predicted under climate change scenarios has resulted in a national imperative to establish likely impacts on fisheries in Australia. Northern Australia is predicted to be affected by changes in rainfall patterns and resultant changes in river flows to the marine environment, increased intensity of cyclones, increased water temperatures, increases in ocean acidification, and altered current patterns (CSIRO 2007). These changes in the marine environment will directly impact on fisheries including modified phenology and physiology, altered ranges and distributions, composition and interactions within communities, and fisheries catch rates (Hobday et al 2008, Munday et al 2008, Halliday et al, 2008, Balston 2009). Critically, most fisheries in northern Australia are deemed to be not well prepared at all for future climate impacts (Hobday et al 2008). For fishery sectors in northern Australia to be able to respond positively and adapt to climate-induced changes on fish stocks there is a need to determine which stocks, and where, when and how they are likely to be affected. Current fisheries management in northern Australia is jurisdiction-based. There is a need for a co-operative approach to developing management policy that can deal with future climate change scenarios. Development of such policy requires consultation with all stakeholder groups. This addresses one of the NCCARP high priority research needs for commercial and recreational fishing, two of FRDC's Strategic Priority R&D Areas (Themes 3 & 4), and priorities for Qld and NT management agencies.

There exists extensive northern Australia biophysical and fisheries data for regional assessment of likely climate change impacts. Data include temperature, salinity, pH, wind, rainfall, upwelling events and river flows. There is a critical need for the collation of existing data sets to determine and document the key environmental drivers for northern Australian fisheries; a key research priority for national, Qld and NT agencies.

6   OBJECTIVES

1. Describe the projected climate-driven changes that are relevant to northern Australian fisheries resources.
2. Assess the potential impacts of climate change on key fisheries and species in northern Australia.
3. Assess current management to identify approaches that are adaptive to potential climate change scenarios.

7   METHODS

7.1   Overview

This project used a structured approach to achieve the ultimate objective of identifying adaptation options for northern Australian fisheries in response to projected climate change. The key underlying framework for this work was the Vulnerability Assessment framework followed by the Inter-Governmental Panel for Climate Change in their global assessment process (Figure 7.1) (Schroter et al. 2004). This framework provided an intuitive and structured approach for determining the potential impacts of climate change on fisheries species (and systems) and their relative level of vulnerability. The framework also provides transparency to stakeholders by incorporating adaptive capacity thereby informing the development of appropriate responses for relevant fisheries stakeholder groups to consider.
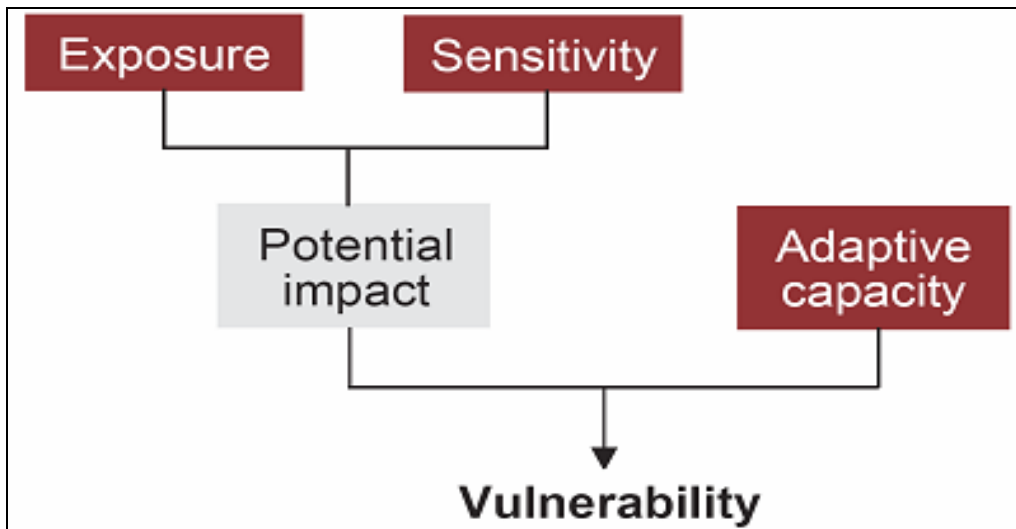
Figure 7.1 Vulnerability assessment framework adopted by the IPCC (Schroter et al. 2004).

For each of the major assessment elements used in this report definitions are as follows:

Exposure:

The nature and degree to which a system or species is exposed to significant climate variations. In a climate change context, it captures the important weather events and patterns that affect the system. Exposure represents the background climate conditions against which a species or system operates, and any changes in those conditions.

Sensitivity:

The degree to which a system or species is affected, either adversely or beneficially, by climate-related stimuli. Climate related stimuli, include mean (i.e. average) climate characteristics, climate variability and the frequency and magnitude of extremes. Sensitive species and systems are highly responsive to climate and can be significantly affected by small changes. Understanding a species or system's sensitivity also requires an understanding of the thresholds at which it begins to exhibit changes in response to climatic influences, whether these adjustments are likely to be 'step changes' or gradual, and the degree to which these changes are reversible. The effect may be direct (eg coral bleaching in response to elevated sea surface temperatures) or indirect (eg loss of suitable habitat for important fisheries species due to changes in sea temperature, ocean chemistry and storm intensity).

Adaptive Capacity:

The potential for a species or system (natural or social) to adapt to climate change (including changes in variability and extremes) so as to maximise fitness, to moderate potential damages, to take advantage of opportunities or to cope with consequences.

Vulnerability:
The degree to which a system or species is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate variation to which a system or species is exposed, its sensitivity, and its adaptive capacity.

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The project followed a number of iterative steps to comprehensively address each of the elements of the assessment framework, while also adopting a stakeholder inclusive approach where appropriate to ensure outputs that were relevant and achievable. These steps were key to meeting the major project objectives, which address several of the framework elements (Exposure, Sensitivity, Adaptive Capacity). The steps taken as part of each of the assessment framework elements are summarised as a flow diagram in Figure 7.2. This flow diagram reveals the overall process followed by the project while the details of each of the tasks follow in subsequent sections of this chapter.
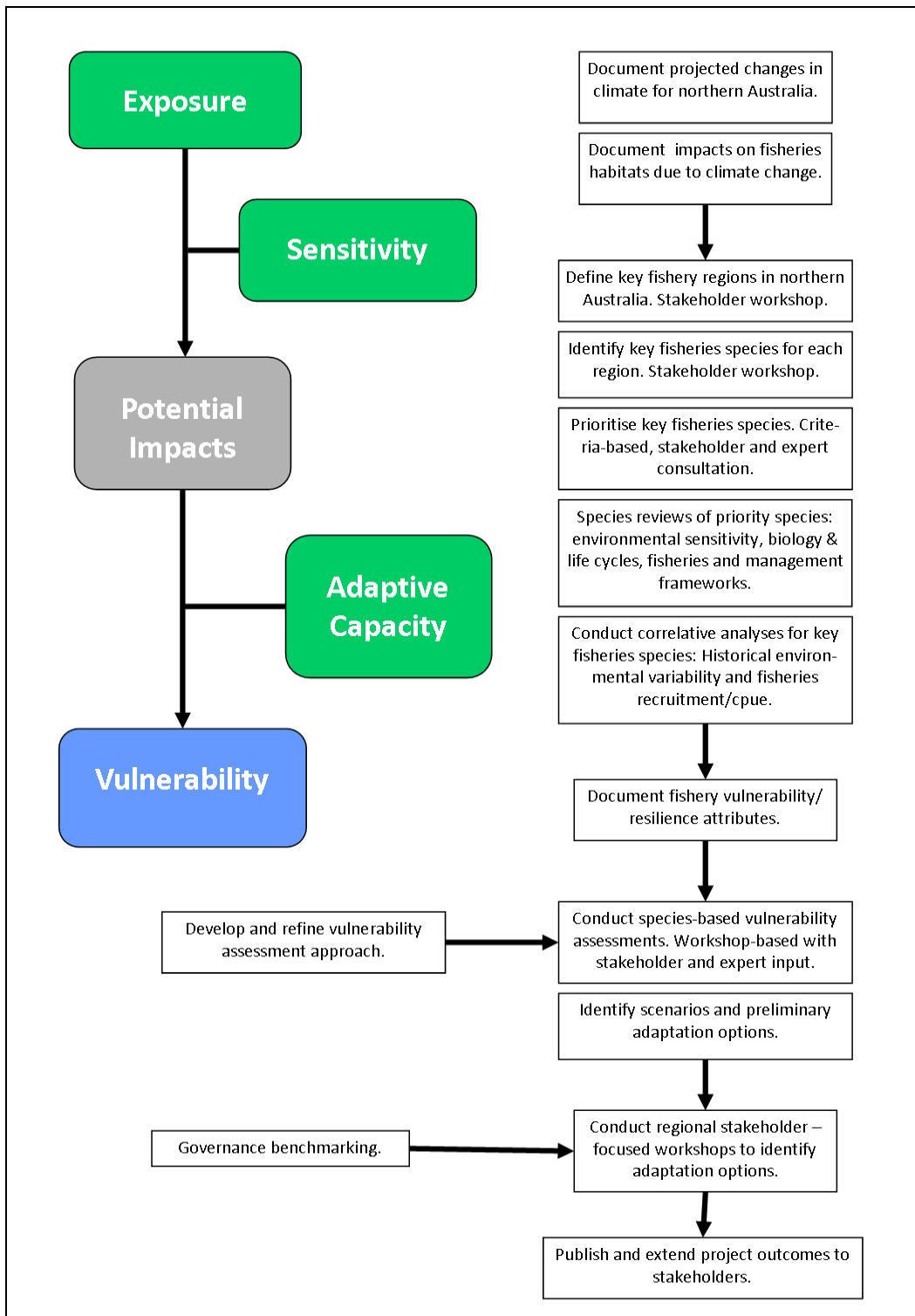
Figure 7.2 Flow diagram of key project tasks carried out in addressing the respective elements of the vulnerability assessment framework.

7.2   Defining geographic scale

The project was focused on fisheries across northern Australia covering a vast area over three jurisdictions. Given the vastness of the total area and the fact that the fisheries and species of importance across this area varies markedly, for the purposes of this project we decided it was appropriate to divide northern Australia into three major fishery regions. The three key regions were: north-western Australia (northern Western Australia and north-western Northern Territory; NWA), the Gulf of Carpentaria (GoC), and the Queensland east coast (EC) (Figure 7.3). The geographic limit of interest on the east and west coasts was determined primarily by the usual ranges of key species with some species extending into New South Wales due to seasonal migrations, with the notable example being Spanish mackerel (*Scomberomorus commerson*).

Figure 7.3 Australian map indicating the key spatial regions adopted for identifying the key northern Australian fisheries species.

7.3   Species identification

To assess the potential impacts on the different fisheries of northern Australia we focused on key species and assessed each species independently. Identification of the key species of interest was initiated at the first project workshop in Brisbane in April, 2011, where the project team in attendance comprised of stakeholders from Queensland and Northern Territory and included scientists, fisheries and conservation managers, commercial fishing

interests, and recreational fishing interests. A full list of participants is provided in Appendix 3. At this workshop we conducted a 'brainstorming' session to list key species and based our species selection on three criteria: fisheries importance (social value, economic value, level of catch), potential sensitivity to climate change and, to a lesser extent, data availability. This initial list was then sent out to a wider reach of stakeholders for comment and addition of new species if necessary. Western Australian fisheries interests were consulted at this time for input into the species list for the NWA region. Once feedback had been received from all stakeholders a final list of key northern species was collated for each of the three regions.

7.4   Species prioritisation

Given the large number of species and the limited time available it was not possible to include all the listed species in the data analyses or, potentially, the project vulnerability assessment stage. Therefore we prioritised species lists for each region using a semi-quantitative framework. This was not intended to produce a definitive ranking of the importance of northern Australian fisheries species, although the final lists would likely be indicative of this. It was however, intended to provide guidance to the project of the species that should receive our focus, and the order in which we would proceed through the list of species in order to provide an assessment of as many northern Australian fisheries species as possible.

The framework was comprised of two "groups" of criteria relating to attributes of each species: Group 1. Fisheries/ecological attributes; and, Group 2. Climate change sensitivity attributes. The criteria used for the Group 1 attributes and their definitions are given in Table 7.1. For each region, project members and other "expert" stakeholders subjectively scored against the criteria for each species for Group 1 using relative scores of 3 (high importance), 2 (medium), and 1 (low). Scorers were chosen based on their expert knowledge of species biology and ecology, and fisheries for the respective regions. For each species, the individual scores for each criterion were summed to give a total Group 1 score. The final Group 1 score for each species was taken as the average score for that species across all scorers.

Group 2 criteria for climate change sensitivity were based on those developed by Pecl et al (2011) in their ecological risk assessment for south-eastern Australian fishery species. Although there are differences in the tropics compared to south-eastern Australia (eg. dramatic episodic disturbances such as cyclones and floods), many of the criteria used in the Pecl et al (2011a) framework capture these issues in some way. These criteria and how they were scored are detailed in Table 7.2. For the Group 2 criteria scientific experts were used for scoring each species and, given the lack of knowledge of species sensitivity to environmental variation, were asked to use their "professional judgement" in scoring and in some cases this meant using "educated guesses". For each species, the mean score of each

of the attributes (Abundance, Distribution and Phenology), were summed to give a total Group 2 score for the individual scoring. The final Group 2 score for each species was taken as the average score across all scorers for that species.

Table 7.1 Criterion used for fisheries/ecological attributes of each species in the semi-quantitative framework used to prioritise species from each region for further analysis.

Criterion	Guiding definitions
Social/cultural importance	Species historically targeted due to popularity as a sportfish, edible qualities, large size, or other historical significance
Economic importance	Dollar value mostly as a commercial target species or through high recreational effort and/or tourism attraction
Catch	Volume of catch
Ecological importance	Considers trophic level and interactions

The final score used for ranking individual species in order of priority was the sum of the mean Group 1 score and the mean Group 2 score. The scoring process described here and above are summarised in Figure 7.4 using east coast barramundi and mud crab as examples.

Table 7.2 Criterion used for climate change sensitivity attributes of each species in the semi-quantitative framework used to prioritise species from each region for further analysis (Pecl et al 2011a).

Sensitivity attribute		Risk category (sensitivity and capacity to respond to change)		
		High sensitivity (3), low capacity to respond	Medium (2)	Low sensitivity (1), high capacity to respond
Abundance	Fecundity – egg production	<100 eggs per year	100 - 20,000 eggs per year	>20,000 eggs per year
	Recruitment period – successful recruitment event that sustains the abundance of the fishery	Highly episodic recruitment event	Occasional and variable recruitment period	Consistent recruitment events every 1-2 years
	Average age at maturity	>10 years	2-10 years	≤2 years
	Generalist vs. Specialist – food and habitat	Reliance on both habitat and prey	Reliance on either habitat or prey	Reliance on neither habitat or prey
Distribution	Capacity for larval dispersal or larval duration – hatching to settlement (benthic species), hatching to yolk sac re-adsorption (pelagic species)	<2 weeks or no larval stage	2 – 8 weeks	>2 months
	Capacity for adult/juvenile movement – lifetime range post-larval stage	<10 km	10 – 1000 km	>1000 km
	Physiological tolerance – latitudinal coverage of adult species as a proxy of environmental tolerance	<10° latitude	10 - 20° latitude	>20° latitude
	Spatial availability of unoccupied habitat for most critical life stage – ability to shift distributional range	No unoccupied habitat; 0 - 2° latitude or longitude	Limited unoccupied habitat; 2 - 6° latitude or longitude	Substantial unoccupied habitat; >6° latitude or longitude
Phenology	Environmental variable as a phenological cue for spawning or breeding – cues include salinity, temperature, currents and freshwater flows	Strong correlation of spawning to environmental variable	Weak correlation of spawning to environmental variable	No apparent correlation of spawning to environmental variable
	Environmental variable as a phenological cue for settlement or metamorphosis	Strong correlation to environmental variable	Weak correlation to environmental variable	No apparent correlation to environmental variable
	Temporal mismatches of life cycle events – duration of spawning, breeding or moulting season	Brief duration; <2 months	Wide duration; 2 – 4 months	Continuous duration; >4 months
	Migration (seasonal or spawning)	Migration is common for the whole population	Migration is common for some of the population	No migration

Group 1	Mud crab	Barramundi
social/cultural importance	3	3
economic importance (GVP)	2	3
catch (net volume)	2	3
ecological importance	3	2
SUM:	10	11

Group 2	Sensitivity attribute	Mud crab	Barramundi
Abundance	Fecundity	2	1
	Recruitment period	3	3
	Average age at maturity	2	2
	Generalist vs Specialist	2	2
	Ave:	2.25	2.00
Distribution	Larval dispersal/duration	1	2
	Adult/juvenile movement	3	2
	Physiological tolerance	2	2
	Spatial availability of unoccupied habitat	2	3
	Ave:	2.00	2.25
Phenology	Environmental spawning cue correlation	3	3
	Environmental settlement cue correlation	2	1
	During of spawning season	2	3
	Migration	1	1
	Ave:	2.00	2.00
	TOTAL:	6.25	6.25

Group 1 criteria	Scorers						
Species	1	2	3	4	5	6	AVE:
Barramundi	10	10	12	12	11	11	11.00
Mud crab	10	12	11	12	11	10	11.00

Group 2 criteria	Scorers				
Species	1	2	3	4	AVE:
Barramundi	6.25	5.25	6	6.25	5.94
Mud crab		4.75	7	6.25	6.00

FINAL RANKINGS:

Species	Group 1	Group 2	Score
Mud crab	11.00	6.00	17.00
Barramundi	11.00	5.94	16.94

Figure 7.4 Summary of the scoring framework used for prioritisation of each species within each region. In this example we used east coast barramundi and mud crab. The two top spreadsheet screen captures show Group scores from an individual expert with the following screen captures showing how all individual scores are collated.

7.5   Species reviews

Based on the prioritised list of fishery species pooled across the three northern Australian regions, we produced detailed species profiles. These profiles were based on the species reviews done by Pecl et al (2011b) and so were comprised of information about the fisheries, their management, biology and life history, as well as documenting known and inferred information about species sensitivity to environmental change. These reviews not only serve as a useful stand-alone resource for all fishery practitioners, now and into the future, but they also provide the necessary baseline information for this project to (i) carry out further species sensitivity analyses, (ii) conduct the species-based vulnerability

assessments, and (iii) identify appropriate adaptation options and barriers. A total of 23 species reviews were compiled and are collated into a companion publication (Part 2) to this vulnerability assessment report.

7.6   Observed and projected climate for northern Australia

7.6.1   Observed Climate

We collated data for observed ocean and surface climate for variables that tropical fisheries are most likely to be sensitive to, based on the sensitivity analysis conducted at the project workshop in December 2011. The information was drawn from a range of sources, particularly the Australian Bureau of Meteorology, CSIRO (CSIRO and BoM 2007) and the Queensland Government as well as key literature, particularly Lough and Hobday 2011, Church et al. 2009 (for sea level) and Lough 2007 (for detailed information on the Great Barrier Reef). Further detailed information for other project regions (e.g. Gulf of Carpentaria, northwest WA) were sourced from state and regional datasets. The summary of observed climate covered historic temporal periods when the data are most reliable.

7.6.2   Climate Projections

The climate projections for this project were compiled from a range of sources including Climate Change in Australia (CSIRO and BoM 2007), OzClim using the CSIRO Mk3.5 model, and SPC 2011 (Table 7.3).

Table 7.3 Climate variables selected for climate projections and data sources.

Variable	Data source
SST	OzClim
Ocean temp 250 m	CSIRO and BoM 2007
Rainfall	CSIRO and BoM 2007
Riverflow	CSIRO and BoM 2007
Ocean pH	SPC 2011
Storms & Cyclones	Ozclim
Sea level	CSIRO and BoM 2007
Ocean circulation	Ozclim

Ultimately, the projections are all based on the outputs of global climate models. A climate model is a numerical description that represents our understanding of the physics, and in some cases chemistry and biology, of the ocean, atmosphere, land surface and ice regions. All models are state-of-the-art 'coupled' models, meaning that ocean, atmosphere, land and ice models are coupled together, with information continuously being exchanged between these components to produce an estimate of global climate. These climate models are run

for hundreds of simulation-years subject to constant, pre-industrial (1870) forcing, i.e. constant solar energy and appropriate greenhouse gas levels to develop a baseline. The 20th century simulations incorporate increasing greenhouse gases in the atmosphere in line with historical emissions and using observed natural forcing (e.g. changes in solar radiation, volcanic eruptions). At the end of the 20th century, projection simulations were carried out based on predefined 'plausible' future emission trajectories.

For this project, we focused on two of these trajectories, corresponding to low (B1) and high 'business as usual' (A1FI) emissions scenarios from the IPCC Special Report on Emissions Scenarios (SRES) (IPCC 2007). These emissions scenarios consider a range of possible future global conditions, including economics, population (growth and distribution), energy technologies, and cultural and social interactions. The models can then simulate the atmosphere and ocean based on these possible futures, and in this project we used projections for the near-term (2030) and long-term (2070).

Some of these models are now available as web-based online tools for generating climate change projections, such as the CSIRO-developed OzClim[1]. OzClim provides an Australian-specific model for 12 variables, eight emission scenarios, three climatic sensitivities and 23 global climate models. This allows users to generate projections of annual, seasonal or monthly average changes in climate for the years 2020 – 2100 (in 5-year increments). This model was selected as it is one of the more recent CSIRO climate models developed for Australia and has reasonable 'skill' in capturing present and past states of the Australian climate system to make projections of what the future might hold.

7.7 Climate change implications for habitats that support northern Australian fisheries

We conducted a literature review to summarise information on documented habitat types and extent in the three regions of northern Australia – EC, GoC and NWA – based on published reports, grey literature and online GIS mapping tools (OzCoasts, Geoscience Australia). The review also documented known sensitivities of these habitats to climate drivers, including results from related projects in adjacent regions (Welch and Johnson 2013, Bell et al. 2011a).

The vulnerability of northern Australian habitats to projected climate change was based largely on a review and synthesis of available vulnerability assessment results. The habitats considered in this project – floodplains, coastal bays and estuaries, seagrass meadows, mangroves and coral reefs – have been assessed in tropical regions using the structured vulnerability assessment framework with the elements of Exposure, Sensitivity and Adaptive Capacity. Therefore, this project reviewed and selected comparable results from habitat

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[1]http://www.csiro.au/ozclim/home.do

vulnerability assessments conducted for habitats in the GBR (Johnson and Marshall 2007), Torres Strait (Welch and Johnson 2013) and the Pacific region (Bell et al. 2011a), particularly nearby Melanesian nations. The assessment of vulnerability was tailored to the three regions of northern Australia, taking into consideration observed and projected climate for the regions, the extent and distribution of habitats, and the scale of the regions.

7.8   Sensitivity data analyses

7.8.1   Identifying species and key variables

Although the species reviews document their sensitivity to particular environmental variables, very little of this information is from published studies and so much of what we "know" about species sensitivity is inferred based on expert knowledge and/or studies on similar species. This project was an opportunity to potentially fill some of these information gaps by investigating the quality and quantity of existing relevant fisheries data, and where suitable, examine data on key tropical fisheries species for correlation with historical environmental data.

It was acknowledged from the outset that, given the often-coarse nature of fisheries and environmental data, identifying strong signals that would signify important relationships would be difficult to achieve. To maximise the likelihood that any data analyses conducted would be able to detect significant relationships if they existed, we adopted a hypothesis-driven approach whereby the most plausible drivers of population dynamics were examined for key aspects of the species life history. This process also considered the quantity and quality of data available and the ranking of each species from the prioritisation process.

To help define the hypotheses of interest for the priority species, we used a semi-quantitative approach for determining the environmental drivers most likely to affect each of the particular species. Drawing on known sensitivity and expert opinion, the framework estimated the likely sensitivity of key aspects of each species life history characteristics to particular environmental variables. The approach used was for experts to assign a "1" for each species characteristic thought likely to be sensitive to changes in each of the particular environmental variable. A "0" was assigned if it was thought to be not sensitive. For each environmental variable these scores were summed to give a value ranging from 0 – 4. The relative level of impact of each environmental variable on each species was determined based on the scores and definitions given in Table 7.4. The species characteristics examined were recruitment, growth, distribution and catchability, while the environmental variables examined were Sea Surface Temperature (SST), rainfall, ocean pH, sea level, salinity, upwelling, nutrients, wind/current, and riverflow. An example of this framework and how it was used is given below in Table 7.5 using Spanish mackerel, while the results for this process for all species examined is provided in Appendix 6.

Table 7.4 Derivation of the estimated level of impact of environmental variables on key species.

Impact level	Impact description	Score
High	Substantial effect	3 - 4
Medium	Some effect	2
Low	No effect or unknown	0 - 1

Table 7.5 Framework for identifying likely environmental drivers of interest for each species where Impact is High (H), Medium (M), or Low (L). Aspects of the species are scored 1 or 0 based on their likely sensitivity to each environmental variable and the likely impact derived as described in Table 6.4. The example shown is for Spanish mackerel.

Spanish mackerel	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	1	0	H
rainfall	1	1	0	0	M
pH	0	0	0	0	L
sea level	0	0	0	0	L
salinity (Sur)	0	0	0	0	L
upwelling	1	1	0	0	M
nutrients	1	1	0	0	M
wind/current	1	0	1	0	M
riverflow	1	1	0	0	M

In the example given, this framework helped to identify that SST was a likely key driver of Spanish mackerel population dynamics (eg. recruitment and distribution) and therefore presented plausible hypotheses that may warrant testing through analyses of data. For our priority species we also assessed the availability of fisheries-dependent and fisheries-independent data, evidence of previous research, and the capacity for the project team to carry out the analyses. Through this process species for analysis were identified and relevant hypotheses were developed.

7.8.2   Data analyses

Based on biological aspects of individual fish species and key drivers of fisheries production, as well as the type of data available, there were two main data analysis approaches used. These were: the examination of recruitment dynamics which directly influences fishery production; and fishery catch rate, which can be influenced by past conditions (recruitment and growth), but can also be influenced by current local environmental conditions (catchability). Selection of explanatory environmental variables for inclusion in the global model was based on an integrative approach suggested by Robins *et al.* (2005). This involved carrying out a detailed review of the life history and life cycle of the species of interest in order to systematically identify a subset of biologically plausible environmental

variables and the lag at which they would most likely affect different life stages (see species reviews in the companion report). This reduced the possibility of obtaining statistically significant correlations without any causal relationship (i.e. Type 1 error); the risk of this was potentially high given the often-large size of the datasets available for analysis.

Environmental data used in the respective analyses are described in each section for the relevant species and their sources. Satellite-derived Sea Surface Temperature (SST) data was sourced from NOAA/NASA Pathfinder version 5.2 with data weekly at 4km resolution aggregated across selected fishery grids or sites. Chlorophyll a. data was median monthly data for selected spatial grids and were sourced from NOAA/NASA and CSIRO Land and Water.

Catch rate analyses

Catch and effort data were obtained from the daily commercial fisheries logbooks submitted to the QDAFF and DPIF. These data were obtained from the specific location for each analysis and were aggregated into annual totals to investigate the correlation of inter-annual trends with environmental factors over the same temporal period. All data were transformed ($\log_{10}(x+1)$) prior to analysis to normalise variances. Correlation analyses and all sub-sets general linear models (GLM, Genstat 2008) were used to explore the potential relationships between catch and effort data and environmental variables. The GLM provided a relative contribution of variables individually or grouped to the variation explained by the model terms.

Year Class Strength analyses

To examine for the influence of environmental factors on recruitment success we used Year Class Strength (YCS) analysis following the methods described by Maceina (1997). To undertake these analyses a time series of age structure data was obtained for each species, where possible, from annual collections of otolith samples. Catch curves were generated for each year of data by taking a weighted linear regression of the natural log of abundance against age for the descending part of the curve. This approach uses positive and negative residuals associated with the linear catch curve as being strong and weak year classes respectively (Maceina 1997). Based on hypotheses for each species, environmental variables can then be examined as predictors of YCS. The relationship between YCS and environmental variables was investigated by correlation analysis and all sub-sets general linear modelling (GenStat 2008) with year-class strength as the response variable, and age and sample year as forced variables because the abundance of individual age-classes is not comparable between years (Staunton Smith et al. 2004).

Both analysis types were investigated for the degree of auto-correlation amongst residuals and, where significant, the degrees of freedom were adjusted to account for serial auto-correlation (Pyper and Peterman 1998).

7.8.2.1 Barramundi

Author: Thor Saunders

The analyses for this species was conducted on data collected from the Daly River, located approximately 200 km to the south of Darwin in the Northern Territory (NT). This river has one of the largest catchments in the NT and is an important area for both commercial and recreational fishers (for a detailed description see Halliday et al. 2012). The specific hypotheses investigated in these analyses were:

1. That increases in river height and rainfall as a proxy for flood plain inundation will increase barramundi catch.
2. That increases in river height and rainfall as a proxy for flood plain inundation will increase larval/early juvenile growth and survival.

Catch Data Analysis

Fishery catch and effort data were obtained from daily logbook records submitted to the Fisheries Division of the NT Department of Primary Industry and Fisheries by both commercial and Fishing Tour Operators (FTOs). Catch was aggregated into annual totals to investigate inter-year trends. Data was available during the periods 1983-2012 and 1994-2012 for the commercial sector and FTO sectors respectively. Catch rate was obtained by dividing annual catch by annual effort and the environmental variables included annual water year (e.g. October 2009 to September 2010 = 2010 water year) rainfall (mm) from Katherine (termed 'rainfall') (Bureau of Meteorology) and river height data in number of days above 10m at the Daly River crossing (termed 'river height') (NT Department of Land Resource Management) over the same period as the catch and effort data.

Correlation coefficients were calculated between annual CPUE and river height and rainfall variables. An all sub-sets general linear model (GLM, Genstat 2008) was used to more thoroughly explore potential relationships between catch and the environmental parameters. Instead of using CPUE as the dependant variable, catch was used and effort and sampling year were forced into the model so that the variation in catch attributed by variation in these variables could be quantified separately. To investigate the temporal influence of river height and rainfall, lags were included as independent variables 1, 2 and 3 years (termed 'river height 1, 2 and 3' and 'rainfall 1, 2 and 3') before the current water year.

Year class strength analysis

The age-structure of the Daly River barramundi population was determined by carrying out opportunistic sampling on commercial and recreational fisher catches throughout each sample year (2007-2011). Each sample had the total body length measured and otoliths were retained for ageing. Because of the selectivity of commercial and recreational gear types it was assumed that only the 3-8 year age classes were sampled representatively. This

allowed the YCS analysis to be conducted during 2001-2009. The environmental variables used were the same as for the correlation analysis above. However, the seasonal influence of rain on catch was investigated by including seasonal (summer, autumn, winter and spring) rainfall as independent variables. River height data was not separated seasonally as flooding events were very consistent in late summer and early autumn each year and so was well represented by the annual total. Age was forced into the model, as was sampling year, because the abundance of individual age-classes is not comparable between years (Staunton Smith et al. 2004).

7.8.2.2 Coral trout

Authors: Andrew J. Tobin, Alastair V. Harry, Richard Saunders and Jeffrey Maynard

In an attempt to begin to better understand some of the processes that may drive the variable recruitment of *P. leopardus* that has been described historically, analyses were conducted to investigate – firstly, the presence of significantly variable year class strength (YCS) throughout a time series of age structure data; and secondly, where significant fluctuations in YCS are detected can these patterns be correlated with environmental variables? Based on a working group and the species review for coral trout, the following environment recruitment hypotheses were proposed and investigated in this analysis:

1. SST may affect recruitment by its influence on timing and duration of spawning and by increasing larval/early juvenile growth rates and thus survival
2. High rainfall in coastal catchments as a proxy for primary productivity may have an influence on larval/early juvenile growth and thus survival
3. Fluctuations in the SOI may affect recruitment indirectly through its influence on SST, rainfall and coastal productivity

Year class strength analysis

Age structure data collected by the CRC Reef Effects of Line Fishing Project were utilised for the analysis of year class strength. This data set incorporated 11 consecutive years of fisheries-independent age data from 1995-2005 for three regions (Storm Cay, Mackay and Townsville) from within the GBR Marine Park (GBRMP). Including region as a factor in the initial YCS analyses was paramount as *P. leopardus* are known to vary in both biology and local abundance throughout the GBRMP (Adams et al 2000; Tobin et al 2013). Details of the ELF project sampling protocols and age estimation procedures are available in Mapstone et al. (2004).

This age structure data were used to derive YCS estimates. This data was derived for each year of sampling, *i,* by using the Studentized residuals from a linear regression of $\log(N_i) = a + bx_i$ to provide replicate estimates of relative abundance in the year, *i - x*. Only fish aged 4-11 were included in the analysis since 0-3 year old fish were not fully recruited to the sampling gear, and fish > 11 years were also excluded because they were relatively rare. To

avoid the confounding effects fishing can have on YCS signals (see Russ et al. 1996) only data from unfished reefs were analysed.

For each region, year class strength estimates were correlated against environmental variables hypothesised to possibly impact recruitment processes (Table 7.6). Again as *P. leopardus* is known to vary in both biology and local abundance throughout the GBRMP, SST data for the correlation analysis was localised to spatial grids corresponding to the reefs sampled in each region (Townsville, Mackay, Storm Cay). In addition, SST data were restricted to the Spring spawning period, also the timing of fish sampling. As the timing of *P. leopardus* spawning varies within latitude, the period of Sep-Nov was chosen for Townsville and the period Oct-Dec for Mackay and Storm Cay. The mechanistic impact of the Southern Oscillation Index (SOI) on year class strength is likely to be very broad, thus SOI was considered as an annual mean.

Data from major river catchments close to the sampled regions were selected as proxies for regional rainfall and catchment inundation. The catchment chosen for Townsville was the Burdekin, and for both Mackay and Storm Cay the Fitzroy River was chosen. These are the largest and most representative of river discharge into the GBRMP within these regions. The final river-flow index was the log-transformed sum of river discharge over the Spring period (Sep-Nov) as well as the Spring and Summer period (Sep-Feb). This time-period encompassed the key spawning period and the following wet-season period across northern Australia.

Each YCS vs environmental variable correlation was examined at three different time steps. Correlations were fitted to the year of interest (e.g. year of recruitment or year of catch) as well as the years immediately prior (one year lag; -1) and the year immediately after (one year in advance; +1) in order to help establish whether environmental correlations had a causal basis.

Table 7.6 Description of environmental predictors investigated for analysis of year-class strength environment-recruitment relationships.

Variable	Description
SST	Annual means for each region (Storm Cay, Mackay, Townsville)
	Spawning period means for each region (Storm Cay, Mackay, Townsville)
SOI annual	Annual mean for the Coral Sea
River flow	Burdekin flow annual total, log transformed (for Townsville)
	Fitzroy flow wet season flow, log transformed (for Mackay and Storm Cay)

7.8.2.3 Golden snapper

Author: Bill Sawynok

Tagging data was used to determine if there has been any shift in the range of golden snapper at the southern end of their range on the east coast of Australia, and assess whether it may be attributable to local estimates of sea surface temperature.

The tagging data was provided from the Suntag program, managed by Infofish Australia, and included all golden snapper tagged since the mid 1980s. Tagging was carried out voluntarily as part of normal fishing trips and data collected included tag number, date, total length and location. Locations are recorded within the database using Suntag Grid Maps that have either 1 or 2km^2 grids. This provided fine scale resolution of where fish were tagged and the opportunity to examine whether this had changed over time. Locations where golden snapper were tagged were examined for the period 1985-2013 and data were aggregated over each 5-year period. Data were also aggregated by latitude within half-degree zones from 22°-26° S as shown in Figure 7.5. Tagging data were further constrained to estuary and nearshore habitats; the habitats that golden snapper mostly use.

In addition, Captag, an ANSAQ club in Rockhampton, with approval from the Department of Defence, tagged fish in the creeks at the southern end of Shoalwater Bay from 2000-2012. Tagging was undertaken on trips involving a maximum of 10 boats for 2-3 days each trip and involved the capture and tagging of many golden snapper (Sawynok 2013). Catch and effort details were collected on all trips making this a consistent dataset. As this effort was significantly different from the normal tagging effort the data were analysed separately. The percentage of golden snapper tagged in the catch for Shoalwater Bay trips was calculated along with the CPUE.
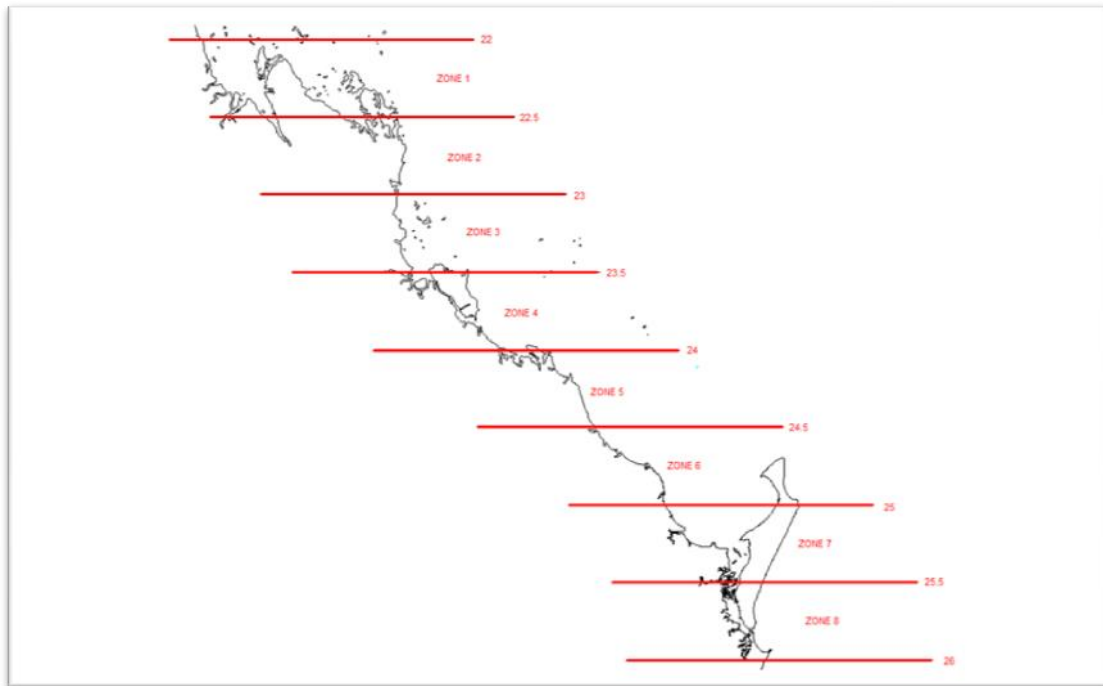
Figure 7.5 Map showing half degrees zones from 22°-26°S on east Australia coast used to analyse golden snapper data.

For each of the eight half degree zones from 22°-26°S the total tagging effort was calculated as the number of days on which fish were tagged in each 5-year period. For each zone and each period the percentage of golden snapper tagged compared with the total fish tagged was calculated. For each zone and each time period the percentage of golden snapper compared with the total tagging effort (number of days on which fish were tagged) was also calculated. Recaptures of golden snapper were examined for the direction of movement and whether movement could be related to season.

It was considered that temperature tolerance levels may be a factor in limiting the range of golden snapper and that any range change may be correlated with any temperature change. Sea surface temperature (SST) data were obtained from NOAA/NASA Pathfinder version 5.2 data at 4 km resolution (nearshore grids only) aggregated across each zone on a seasonal basis from 1985-2009. From that data the mean SST for each zone and each season were calculated.

7.8.2.4 Red throat emperor

Authors: Richard Saunders, Alastair V. Harry, Andrew J. Tobin and Jeffrey Maynard

These analyses focused on the Queensland east coast red throat emperor population and used age structure data collected during the CRC Reef Effects of Line Fishing (ELF) Project (Mapstone et al. 2004). During this project red throat emperor were sampled using commercial fishing gear using fishery-independent methods from three regions on the Great Barrier Reef (Storm Cay, Mackay and Townsville) over a period of eleven years (1995-2005). This provided the basis for a time series of age structures to be constructed for each region. Details of the ELF project sampling protocols and age estimation procedures are available in Mapstone et al. (2004). Based on a working group and the species review for red throat emperor (see Part 2 companion report), the following environment recruitment hypotheses were proposed and investigated in this analysis:

1. SST may affect recruitment by its influence on timing and duration of spawning and by increasing larval/early juvenile growth rates and thus survival
2. High rainfall in coastal catchments as a proxy for primary productivity may have an influence on larval/early juvenile growth and thus survival
3. Fluctuations in the SOI may affect recruitment indirectly through its influence on SST, rainfall and coastal productivity

Year class strength analysis

Age structures were provided from the ELF data sets (Mapstone et al. 2004). This data set was used to estimate year class strength estimates using the methods of Maciena (1997). These estimates were derived for each year of sampling, *i,* by using the Studentized residuals from a linear regression of $\log(N_i) = a+bx_i$ to provide replicate estimates of relative abundance in the year, *i - x*. Only fish aged 6-12 were included in the analysis since 0-5 year old fish were not fully selected by the sampling gear, and fish > 12 years were excluded as they were relatively rare. To avoid the confounding effects fishing can have on YCS signals only data from sanctuary zones (unfished) were considered in these analyses.

The final data treatment necessitated pooling data across regions and providing year class strength estimates for the Queensland east coast (see results and discussion) and correlating these estimates to environmental variables that are likely to act on a broad spatial scale (Table 7.7). Thus, SST data for the correlation analysis was the Coral Sea mean. The SST data were restricted to Spring (Sep-Nov) when spawning activity peaks (Williams et al. 2006), Spring and Summer (Sep to Feb) (for cumulative impact across different life stages), and an annual average. The mechanistic impact of the Southern Oscillation Index on year class strength is likely to be very broad, thus SOI was considered as an annual mean.

River flow data from a major river catchment close to the sampling locations were selected as a proxy for regional rainfall and catchment inundation. The catchment chosen was the Burdekin as the largest and most representative of river discharge to the Great Barrier Reef. The final river-flow index was the log-transformed sum of river discharge over the Spring period (Sep-Nov) as well as the Spring and Summer period (Sep-Feb). This time-period encompassed the key spawning period and the following wet-season period across northern Australia.

Similar to analyses for coral trout, each YCS vs environmental variable correlation was examined at three different time steps. Correlations were fitted to the year of interest (e.g. year of recruitment or year of catch) as well as the years immediately prior (one year lag) and the year immediately after (one year in advance) in order to help establish whether environmental correlations had a causal basis.

Table 7.7 Description of environmental predictors investigated for analysis of year-class strength environment-recruitment relationships.

Variable	Description
SST	Coral Sea mean for full calendar year
	Coral Sea mean for spring period
	Coral Sea mean for Spring and summer period
SOI annual	Annual mean
River flow	Burdekin flow annual total, log transformed
	Burdekin flow wet season flow, log transformed

7.8.2.5 Saucer scallops

Author: Julie Robins

The analyses were conducted in the area between 22°30′ S, 151° E and 26° S, 153°30′ E on the Queensland east coast where the majority of saucer scallops (*Amusium japonicum balloti*) are harvested in Queensland. It includes the inner shelf of the Great Barrier Reef on the Capricorn Coast approximately from Cape Clinton southwards to Hervey Bay. Waters in this area are generally less than 50 m deep and have a high sand content (Pitcher et al. 2007). The area is southwest of the Capricorn Channel and west of the Capricorn Bunker Group of coral cay islands. In the Capricorn region, these islands separate the inner shelf of the GBR from deeper (>200m) waters (Burrage et al. 1996). Mesoscale eddies of the East Australian Current have been reported in the area (Burrage et al. 1996) and are usually cold core cyclonic eddies. The study area also includes the relatively nearshore areas immediately east of Fraser Island, which intermittently have significant catches of saucer scallops.

From the species profile of the saucer scallop and its fishery (see species reviews in the companion report) the following environment recruitment hypotheses were proposed:

1. Water temperature may affect recruitment by its influence on gonad size and subsequent gamete production (in February and March) by benthic mature scallops.
2. Water temperatures may affect recruitment by its influence on mortality during pelagic larval phases between June and November.
3. Chlorophyll-a., as a proxy for food availability, may influence the larval growth and thus survival between June and November, subsequently affecting spatfall and recruitment. Timing of larval phase is between June and November.
4. Chlorophyll-a., as a proxy for food availability, may affect the growth and survival of benthic juvenile saucer scallops. The benthic juvenile phase occurs between September and December, with the duration of the juvenile phase is likely to vary depending on growth rates, which may also be linked to water temperatures.
5. Hydrographic features, such as cold core eddies, in the area between 22° S and 25° S, may have an influence on the distribution of larvae (June to November) and subsequently affect spatfall by entraining larvae to "optimum" areas.
6. Water temperatures (between September and December) may affect the growth rates of juvenile scallops and therefore the timing of scallop recruitment to the fishery at the legal size limit.
7. Large discharge from adjacent coastal rivers may impact negatively on scallop "recruitment" (Morison and Pears 2012) through reduced salinities or increased turbidity.

Spatial Recruitment Index

Indices of scallop abundance were provided by Dr Alex Campbell from the Queensland Department of Agriculture, Fisheries and Forestry (QDAFF). These indices represent the average scallop density of 0+ and 1+ year old scallops in 43 spatial cells across the study area, based on the scallop fishery independent surveys conducted annually in October between 1997 and 2006. For further details see Campbell *et al.* (2011). The spatial recruitment index data is standardised for sampling and fishing power differences over the duration of the LTMP scallop surveys. Data were available for 43 spatial cells, but only spatial cells where sampling occurred for ≥7 years were included in the analysis. This provided 26 spatial cells with a time series of fishery-independent abundance between 1997 and 2006 (i.e., cells 2-9 occurring within CFISH Grid V32; cells 10-18 occurring within CFISH grid T30; cells 21-28 occurring within CFISH Grid S28, and cell 43 occurring within CFISH Grid R28).

Commercial catch data

Commercial scallop catch and effort data were obtained from Queensland Department of Agriculture, Fisheries and Forestry (QDAFF). A subset of the commercial catch data was used

in the analysis as an index of scallop abundance. The subset included detailed information on effort creep parameters that are not available for the full commercial catch dataset and focuses on the area between 22°30' S, 151° E and 26° S i.e., the main scallop grounds. This subset data is updated annually and used by QDAFF in the catch rate standardisation procedure and fishing power analysis for Queensland saucer scallops. For further details see O'Neill and Leigh (2006) and Campbell *et al*. (2010). The data included daily catch weight (standardised in baskets) and effort (hours trawled) per boat per CFISH grid. Additional effort creep information included: otterboards, presence of a BRD and or TED; presence of a GPS; engine horse power; lunar phase and lunar phase advanced; net size; presence of a kortz nozzle; catch weight (kg) of prawns; presence of sonar devices; trawling speed; and presence of a try net. Commercial catch and effort data were available from 01/01/1988 to the 31/12/2011 and provided 102,355 daily records of catch per boat. The data was filtered to remove records where: (i) hours trawled per day exceeded 24 (i.e., bulk data); and (ii) prawn catch exceed 50 kg (i.e., not targeting scallops); thus providing ~91,000 daily records of catch and effort information. Queensland scallop data were aggregated into fishing years, reflecting biological characteristics of the species and operational characteristics of the fishery (O'Neill and Leigh 2006). For Queensland saucer scallop the fishing year is November (in the preceding year) to October i.e., FishYear 1989 = November 1988 to October 1989.

Environmental factors included in the analysis were selected on the basis that they: (i) were ecologically relevant; (ii) had available data; and (iii) were as close to the biological process/hypothesis as possible (Dormann et al. 2013).

SST data was used as a weekly median SST per 30' x 30' CFISH grids for the time series between January 1986 and December 2011. SST data were explored to investigate the most appropriate ways of aggregating SST data to capture its potential influence on the life history of saucer scallops. These included:

      (i) the median seasonal weekly SST per grid for Summer (Dec to Feb: SST Sum), Autumn (Mar to May: SST Aut), Winter (Jun to Aug: SST Win) and Spring (Sep to Nov:SST Spr);

      (ii) the number of days (wks x 7) per year when the weekly SST across all grids in the study area was <25°C (SST Days25 May to Nov);

      (iii) the median weekly SST across all grids between May and November inclusive (SST May to Nov);

      (iv) the median weekly SST across all grids, when SST was <25°C (Median SST 25);

      (v) the minimum weekly median SST across all grids between May and November (Min SST Nov to May); and

      (vi) the average winter (June to August) weekly median SST across all grids (SST Win all grids).

Although numerous measures of SST were explored (e.g. mean annual), further analyses used the median seasonal SST per grid to encompass spatial and temporal variability that matched commercial catch.

Chlorophyll a was included as median monthly interpretations of Chlorophyll a per CFISH grid were derived from NOAA with CSIRO corrections for GBR waters and were available as a time series between July 2002 and June 2012. Further analyses used the median seasonal SST per grid for Summer (Dec to Feb: Chla Sum), Autumn (Mar to May: Chla Aut), Winter (Jun to Aug: Chla Win) and Spring (Sep to Nov: Chla Spr).

River discharge from the following rivers was assumed to influence the following CFISH spatial grids:
- Fitzroy River – S28, S29;
- Boyne + Calliope River – S30;
- Kolan + Burnett Rivers – U31, U32, T30, T31;
- Mary River – V32, W32;
- Brisbane River – W33, W34, W35: and
- No associated river (i.e., offshore) – T28, T29, U30, V31.

For some areas, the discharge of two rivers was combined because of the close proximity of their estuaries and an inability to separate the area influenced by each river. The discharge of the Burnett and Kolan Rivers were pooled, as was the discharge of the Boyne and Calliope Rivers. Daily river discharge varies by several orders of magnitude between rivers, reflecting the catchment area of each river. Therefore, river discharge was standardised by the respective catchment area (upstream of the gauging station) to make the discharge volumes comparable.

River discharge (i.e., ML day^{-1}) was collated for the most downstream gauging station (GS) between January 1985 and September 2012, available from the Queensland Government Water Monitoring Portal (http://watermonitoring.derm.qld.gov.au/host.htm) for the following rivers:
- Fitzroy River (GS13005A @ The Gap; -23.08897222° S, 150.10713889° S; catchment area = 135,800 km²).
- Calliope River (132001A @ Castlehope; -23.98498333° S, 151.09756389° E; catchment area = 1,288 km²);
- Boyne River (GS133005A @ Awoonga Dam Headwater; -24.07008611° S, 151.32162528° E; catchment area = 2,258 km²);
- Kolan River (GS 135002A @ Springfield; -24.75334711° S, 151.58717235° E; catchment area = 551 km²);
- Burnett River (GS136001 B @ Walla; -25.13617187° S, 151.98222776° E till February 1998 then adjusted flow from GS136007A @ Figtree; -25.28507017° E, 151.9894613° S;

adjustment based on linear regression of flow at GS136007A with flow at GS136001B between January 1997 to February 1998 as recommended by R. Maynard Supervising Hydrographer, DNRM; catchment area = 32,070 km²);

- Mary River (GS 138014A @ Home Park; -25.76832547° S, 152.5273595° E; catchment area = 4,755 km²); and the
- Brisbane River (GS 143001C @ Savages Crossing; -27.43916667° S 152.6686° E; catchment area = 10,170 km²).

Discharge data were aggregated to reflect potential flow impacts on various aspects of scallop biology: January to May for impacts on gonad development (Flow Gonads); June to October for impacts on spawning (Flow Spawn); July to November for impacts on Spatfall (Flow Spat) and November to May for impacts on juvenile growth (Flow Juv).

There has long been anecdotal speculation that oceanographic eddies in the Capricorn area play a key role in influencing saucer scallop abundance. In this area of the Queensland east coast (i.e., between 22° S and 25° S in waters < 500 m deep), mesoscale eddies form as the East Australian Current passes the Swains Reef complex and Capricorn Channel (Weeks et al. 2010). These cyclonic eddies may influence the distribution of pelagic scallop larvae (June to November) and affect the distribution of spatfall and therefore successful recruitment to "optimum" habitat. While the intermittent presence of a cyclonic eddy in the Capricorn area has been documented (Griffin *et al.* 1987; Burrage *et al.* 1996; Weeks *et al.* 2010), no time-series of eddy presence is available. Further quantitative work on the longitudinal duration, spatial extent and intensity of cyclonic eddies in the Capricorn region, particularly when SST are less than 25°C, would be useful in understanding the environmental drivers of Queensland saucer scallop populations. In the absence of detailed longitudinal studies of the oceanography of the Capricorn area, and with advice from Richard Brinkman (AIMS), a surrogate of eddy presence was derived using the available data on the IMOS Ocean Current webpage (http://oceancurrent.imos.org.au/). The daily presence or absence (coded as 1 or 0 respectively) of a cyclonic eddy in the Capricorn area was visually assessed using the available maps of "sea level, geostrophic current and 3-day average SST 1993-now" for the North East (NE) region" (http://oceancurrent.imos.org.au/NE/). Where available, daily eddy presence was confirmed against images in "geostrophic current, snapshot SST + radar 2004-now" for the Southern Great Barrier Reef region (SGBR) (http://oceancurrent.imos.org.au/SGBR/) and Brisbane region (http://oceancurrent.imos.org.au/Brisbane/).

Daily counts of eddy presence between October 1993 and October 2011 were computed into the total number of days per year a cyclonic eddy was visibly present in the Capricorn area (i.e., north west of Sandy Cape) between the months of May and October inclusive. Annual counts were split into two groups: (i) May to July and (ii) August to October to

investigate whether the timing of eddy presence had an influence on early or late spawning respectively.

Analysis of data

Data were transformed (ln(X+1)) prior to analysis to normalise the variances. Correlation coefficients were calculated between environmental factors. Then, all sub-sets generalized linear models (GLM's) were used to explore potential relationships between scallop abundance and environmental factors (GenStat 2011). All-subsets General Linear Model (GLM) will identify the model that explains the greatest amount of variance (as per step-forward GLM) but also calculates all possible combinations of forced and independent factors to identify a number of alternative regression models that can be evaluated by their explanatory power (adjusted R^2) and biological plausibility.

7.8.2.6 Spanish mackerel

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Introduction

Many characteristics of narrow-barred Spanish mackerel and their fisheries are known to be closely associated with environmental conditions, potentially suggestive of a high sensitivity to climate change in this species (Welch *et al* this publication). From an ecological perspective, temperature is thought to be an important cue for spawning in Spanish mackerel. Additionally, larval transport from outer-shelf spawning reefs to suitable settlement areas in estuaries may be affected by currents and prevailing winds. Importantly, early-life history and survival are also presumed to be influenced by the conditions in estuaries where larvae settle and spend the first few months of life before they begin to disperse offshore.

Reflecting their apparent sensitivity to temperature, commercial fisheries for Spanish mackerel are highly seasonal. For instance, off eastern Australia Spanish mackerel occur over a 20° latitudinal gradient between the Torres Straits (10°S) and Coffs Harbour (30°S). Temporal peaks in landings vary regionally and have anecdotally been associated with the location of the 24°C sea surface temperature (SST) isotherm which is thought to influence the longshore movement and migration of older fish. The majority of the landings in the eastern Australia fishery are from fishers targeting spawning aggregations of Spanish mackerel between Lizard Island and Townsville, thus the timing and duration of these fisheries each year is directly linked to spawning behaviour which is influenced by temperature. These life history traits of Spanish mackerel led McPherson (1981) to speculate that fluctuations in fishery landings may be attributed to the effects of the el

Nino-Southern Oscillation (ENSO), and that this could provide a major challenge for managing the fishery.

Despite considerable interest and speculation there has been little formal investigation of the environment-recruitment links in Spanish mackerel. A clearer understanding of these could be beneficial to management of this commercially and recreationally importance species since, to date, there has been little explicit consideration of environmental variation in stock-assessments. Understanding how environmental variability affects the fishery may help in adapting to future climate-driven changes and in the interpretation of historical change in the fishery. We compared and contrasted the relationships between four key environmental variables (sea surface temperature (SST), Southern Oscillation Index (SOI), coastal rainfall, and primary productivity (Chl-a)) and two indices of abundance (year-class-strength and standardised catch per unit effort) on Queensland east coast *S. commerson*.

Study species and area
This analysis focused on the Queensland east coast Spanish mackerel population and the commercial line fishery that targets the species (Welch *et al* this publication). The stock structure of Spanish mackerel across northern Australia has been well-studied and fish on the east coast, which form a distinct genetic stock, are treated as a single unit for management purposes (Buckworth, Newman *et al.* 2007). While there is considerable genetic homogeneity within the east coast population, inferences from both parasites and otolith microchemistry provide strong evidence of subdivision within stocks and much finer-scale population structure (Moore *et al.* 2003; Newman *et al.* 2009). This seems to indicate that adults are relatively sedentary and exhibit little mixing on scales of 100-300km. Buckworth *et al.* (2007) suggests a metapopulation model as the most likely form of stock structuring. Nonetheless, a proportion of fish migrate longer distances, and on the east coast larger fish appear to migrate seasonally into northern NSW waters. Although fishing occurs around the entire Queensland coast, the majority of Spanish mackerel on the east coast is landed during Spring (Sep-Nov) around the outer-shelf reefs north of Townsville where fish aggregate to spawn (Figure 7.6). Although fish are thought to spawn all along the east coast, the extent to which the Townsville region is the dominant or sole area important for spawning is not known. Historical reports of the fishery indicated that similar spawning aggregations may have previously extended much further north to Lizard Island.
For stock assessment purposes the fishery is divided into five nominal regions; North, Townsville, Mackay, Rockhampton and South (Figure 7.6) (Campbell et al. 2012). These regions were adopted in this study.

Abundance indicators
Two indicators of abundance were used in the present study; catch curve-based year class strength (YCS) and standardised annual catch-per-unit-effort (CPUE).

Year class strength

YCS was determined from annual monitoring of commercial and recreational landings of *S. commerson* by the Queensland Government Department of Agriculture Fisheries and Forestry (QDAFF) from 2001-2011. Representative data on the length structure of fisheries landings were collected from 11 geographic regions on the east coast of Queensland. A subsample of measured fish were then aged, and an age length key (ALK) generated and used to convert the resulting length structure to age.
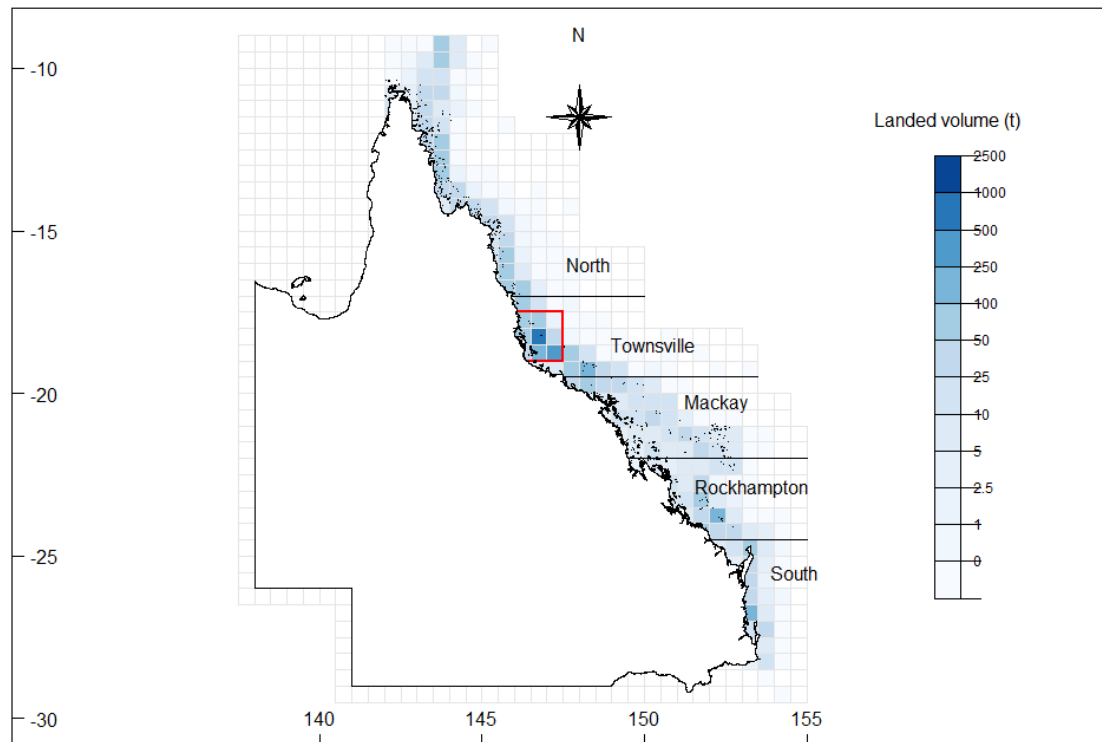
Figure 7.6 Landed-volume (t) of Spanish mackerel, *Scomberomorus commerson*, in east coast Queensland waters 1988–2012 in 0.5×0.5° grids. Five nominal stock assessment regions referred to in the YCS analysis are denoted by the solid black line. The red box denotes the spatial grids used in the CPUE analysis.

The age of subsampled fish was determined by counting growth increments visible on the otolith and based on the otolith edge classification. Because of the highly seasonal nature of the fishery and rapid growth of *S. commerson* a number of age and length adjustments were made prior to constructing the ALK. This was done to ensure that the lengths of fish were comparable throughout the year, and so that fish from the same cohort were kept together. The birth-date for east coast *S. commerson* was assumed to be 1st October with growth bands formed in winter. Individuals sampled between April and August with a 'new' edge classification had one year subtracted to keep them in the previous years' cohort. Individuals with an intermediate or wide edge type between June and October had one year added. Length adjustments were made based on the time between the assigned birth date

and capture. The growth expected during this period of time was added (or subtracted) to the measured length. Length adjustments were based on male and female specific growth models, or a combined model if sex was not determined. The resultant whole age and adjusted length were used to generate ALKs for each sampling year, with both sexes combined (Isermann and Knight 2005).

YCS was measured retrospectively from the resulting age-structure using the catch curve residual method of Maceina (1997). For each year of sampling, *i*, the Studentized residuals from a multiple linear regression of abundance, *N* against age, *x,* and sample year, *i;*

$$\ln(N_i) = a + bx_i + c_i$$

were used to provide replicate estimates of relative abundance of recruits in the year, *i - x*. Individuals younger than two were excluded from the analysis as they were not fully recruited into the fishery and fish older than 11 were excluded because they were relatively rare (Figure 7.7). Linear regression was weighted by sample size (*N*) to minimise the effect of outliers on the analysis (Maceina 1997). For the first year of sampling the residual from the oldest cohort was not used since there was only a single replicate for that year based on only a small number of fish. The coefficient of determination, r^2, from the above regression model was used as a measure of the relative variability in recruitment, with lower values indicative of less stable recruitment (Maceina 1997).

Catch per unit effort
Analysis of CPUE was based on logbook data collected from commercial fishers, who are required to record daily catch of *S. commerson* (weight in kg, or numbers of fish then converted to weight), allocated to a 0.5° spatial grid (Figure 7.6). Although data are collected by fishers along the entire coast, analysis of CPUE was restricted to data collected from the 3×3 array of spatial grids encompassing the principle spawning reefs near Townsville where the majority of fishing occurs in Spring and Summer: $(i, j, k)^T \cdot (18,19,20)$ (Figure 7.6). The data were then limited to a further set of 50 fishers that had landed >25 tonnes of Spanish mackerel since logbook reporting began and who had an average catch >20kg.day^{-1}. These criteria were chosen to limit logbook data to fishers with a long history of specifically targeting Spanish mackerel.
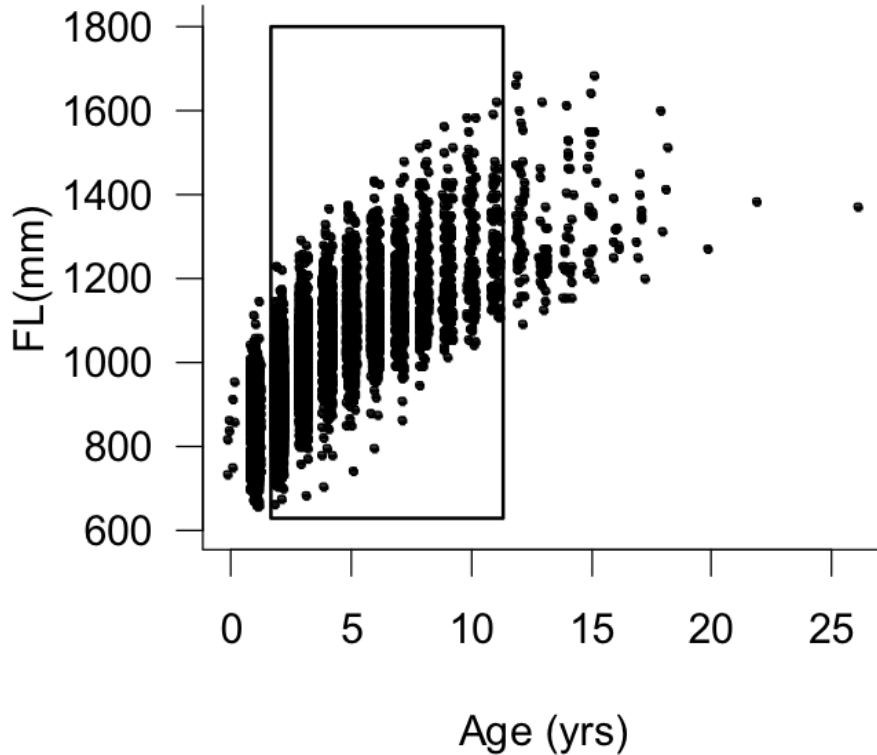
Figure 7.7 Length-at-age data (jittered) available for *S. commerson*. Black box shows ages included in the year-class-strength analysis.

Linear mixed-models were used to standardise logbook data (Maunder and Punt 2004) and followed a similar approach to previous standardisations of Queensland *S. commerson* CPUE data (Begg et al. 2006, Campbell et al. 2012). CPUE standardisation was carried out using a linear mixed effects model and the nlme package in R. The fundamental assumption of this analysis is that the observed catch, *C,* is proportional to the product of effort, *E*, abundance, *N,* and catchability, *q,* (Maunder and Punt 2004), $C = qEN$, such that

$$\ln\left(\frac{C}{E}\right) = \ln(q) + \ln(N)$$, where *q* is a product of fixed and random variables estimated using a mixed effects model (Pinheiro and bates 2000). Financial year (July-June), month, lunar phase and statistical reporting grid were included as fixed variables, and vessel ID was included as a random variable (see Appendix 9) (Begg et al. 2006, Campbell et al. 2012). The estimated year coefficients were extracted and used as the annual index of abundance as

$$\exp\left(\hat{\alpha}_t + \frac{\hat{\sigma}_t}{2}\right)$$, where $\hat{\alpha}_t$ is the estimate of the year coefficient for year *t* and $\hat{\sigma}_t$ is the standard error of $\hat{\alpha}_t$ (Maunder and Punt 2004). Finally standardised year coefficients were divided by the first year of sampling, $\hat{\alpha}_0$.

Interpretation and comparison of abundance indices
The catch-curve residual YCS approach of Maceina (1997) estimates the strength of an individual year class based on the residual of the catch curve corresponding to that year. Because a single year provides an estimate of YCS for many cohorts, multiple years of

sampling can be used to refine estimates. Importantly though, the method only provides a relative index of recruitment and does not reflect the true magnitude of variation in recruitment, overestimating the strength of weak year classes and underestimating the strength of strong year classes (Catlano et al. 2009). Catch-curve derived estimates of YCS are likely to provide a reasonable proxy for YCS providing that recruitment variation exceeds 50-80%, however below this level any variation is likely to be obscured by many factors (Catlano et al. 2009, Tetzlaff et al. 2011). The method is also particularly sensitive to changes in fishing mortality (Catlano et al. 2009).

CPUE was interpreted as index of abundance of mature biomass. The minimum commercial size limit for *S. commerson* in Queensland waters is 75cm, below the length at maturity of this species (~88cm). However, full recruitment to the commercial line fishery typically occurs at age 2, which corresponds to the age at maturity of *S. commerson,* thus the vast majority of fish captured are adults (Figure 7.7).

The two indices of abundance used here are both indirect and representative of different quantities; one is a measure of recruitment and the abundance. To assess their similarity to each other YCS and CPUE (lagged 2 years to account for time to recruit to the fishery) were compared using Pearson's correlation coefficient.

Environmental correlations with recruitment, stock abundance and catchability
Based on an expert working group and the species review (see Part 2 companion report), the following hypotheses were investigated in this analysis:
1. SST may influence on timing and duration of spawning and increasing larval/early juvenile growth rates and thus survival (recruitment). SST may increase feeding activity (catchability) and increasing growth of individuals and population biomass (abundance).
2. Chlorophyll-a as a proxy for primary productivity may have an influence on larval/early juvenile growth and thus survival in estuarine and coastal areas by increasing food availability (recruitment).
3. High rainfall in coastal catchments as a proxy for primary productivity may have an influence on larval/early juvenile growth and thus survival in estuarine and coastal areas by increasing food availability (recruitment).
4. Fluctuations in the SOI are related to changes in SST, rainfall and coastal productivity (recruitment and abundance). SOI may have a general effect on weather conditions (catchability).

The above hypotheses were tested using linear regression analysis; YCS was to investigate environment-recruitment hypotheses and CPUE to investigate stock abundance and catchability hypotheses. YCS data were pooled to match stock assessment regions and were available from each of the broad-scale regions except North (Figure 7.6). In each of the four

regions, SST data for the correlation analysis were selected from a spatial grid in the region of highest catch, or in the Mackay region when that data was not available from an adjacent grid (Table 7.8, Figure 7.8). SST data were restricted to Spring (Sep-Nov) when spawning activity peaks. Any effect of Chl-a was thought to be related to increases in primary productivity of the coastal and estuarine areas important to juveniles. For Townsville and Rockhampton, where key catch grids were further offshore, an adjacent inshore grid was selected for Chl-a. These data were unavailable for Mackay, so the closest adjacent grid was selected. In the South, the highest catch occurred in a coastal grid, so this was also selected for Chl-a. In each region, data from a major river catchment close to the key catch grids was selected as a proxy for regional rainfall and catchment inundation. Catchments chosen were Herbert (Townsville), Burdekin (Mackay), Fitzroy (Rockhampton) and the Brisbane (South). The final river-flow index was the log-transformed sum of river discharge over the Spring and Summer period (Sep-Feb). This time-period encompassed the key spawning period and the following wet-season period across northern Australia.
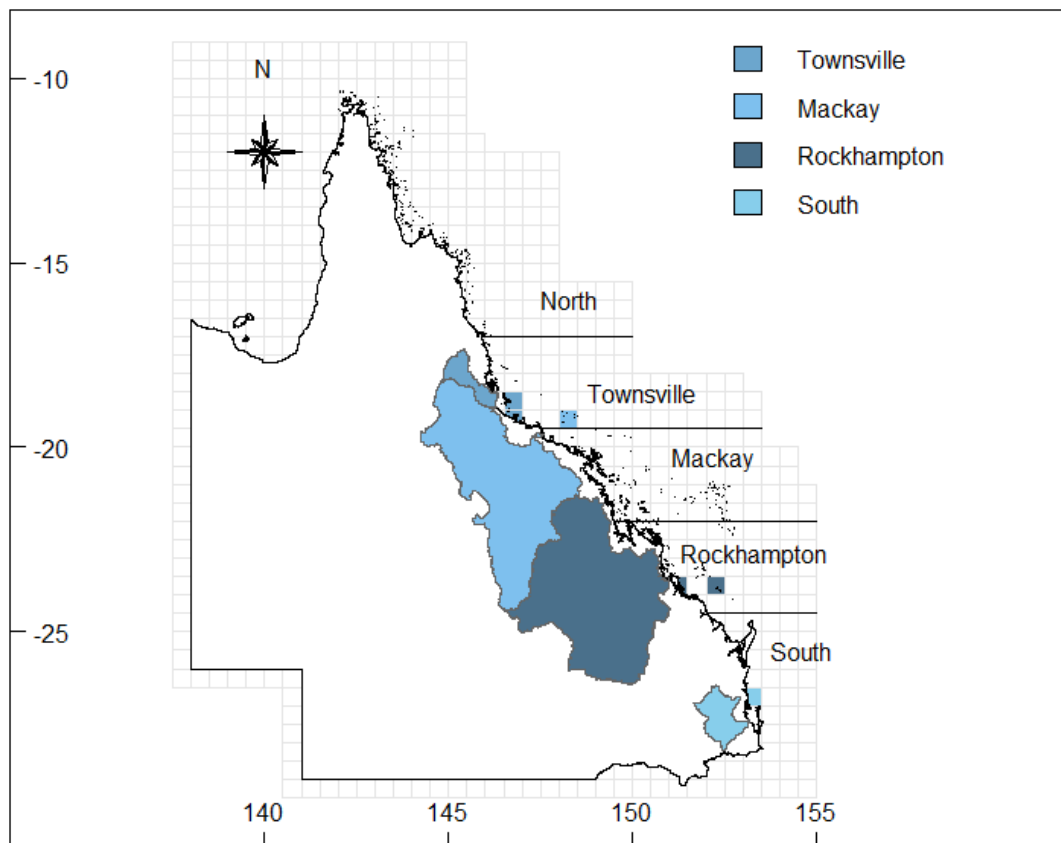
Figure 7.8 Map of the study area indicating regions used in the year-class-strength analysis and catch grids where SST and Chl-a data were sourced. River catchments used in analyses are also highlighted.

It was not immediately possible to separate stock-abundance and catchability hypotheses which were tested simultaneously using CPUE data. Since CPUE data also came from the Townsville region (Figure 7.6), similar environmental data were used, although Chl-a data

was from an offshore grid rather than an inshore grid reflecting differences in the stock-abundance hypothesis (Table 7.9).

Standard bivariate linear regression models were used for all analyses, although a range of models were initially trialled in the YCS analyses. Mixed effects models including a random intercept term and both a random slope and intercept term were also trialled however did not change the results substantially. Correlations were fitted to the year of interest (e.g. year of recruitment or year of catch) as well as the years immediately prior (one year lag) and the year immediately after (one year in advance) in order to help establish whether environmental correlations had a causal basis.

Table 7.8 Description of environmental predictors investigated in each region for analysis of year-class-strength environment-recruitment relationships.

Variable	Townsville	Mackay	Rockhampton	South
SST	Spring mean, J20 (146.75°E, 18.75°S)	Spring mean, M21 (148.25°E, 19.25°S)	Spring mean, U30 (152.25°E, 23.75°S)	Spring mean, W36 (153.25°E, 26.75°S)
SOI	Annual mean	Annual mean	Annual mean	Annual mean
Chl-a	Spring mean in adjacent inshore areas, J21 (146.75°E, 19.25°S)	Spring mean offshore (inshore not available), M21 (148.25°E, 19.25°S)	Spring mean in adjacent inshore area, S30 (151.25°E, 23.75°S)	Spring mean, inshore grid, W36 (153.25°E, 26.75°S)
River flow	Spring and Summer flow, Herbert River, log transformed	Spring and Summer flow, Burdekin River, log transformed	Sum of Spring and Summer flow, Fitzroy, log transformed	Sum of Spring and Summer flow, Brisbane River, log transformed

Table 7.9 Description of environmental predictors investigated for analysis of environment-stock abundance and -catchability relationships

Variable	Description
SST	Spring mean, J20 (146.75°E, 18.75°S)
SOI	Annual mean
Chl-a	Spring mean at spawning aggregation, J20 (146.75°E, 18.75°S)
River flow	Spring and Summer flow, Herbert River, log transformed

7.8.2.7 Summary of analyses

Table 7.10 Summary of the species analysed, the analyses conducted and for which regions, the environmental variables used in each analysis, and sources of all potential data.

Species	Analyses	Regions	Environmental Variables	Data sources
Barramundi	YCS	NT	Rainfall, River height	NT Fisheries, NT DLRM
	Catch rate	NT	Rainfall, river height	NT Fisheries, NT DLRM
Coral trout	YCS	EC; Townsville, Mackay, Storm Cay	SST, SOI, river flow	ELF, CSIRO, NOAA, DERM
Golden snapper	Range shift	Rockhampton	SST	Infofish Australia; NOAA/NASA
Red throat emperor	YCS	EC; Townsville, Mackay, Storm Cay	SST, SOI, river flow	ELF, CSIRO, NOAA, DERM
Saucer scallop	Recruitment	SE Queensland	SST, Chl a., river discharge, eddy currents	NOAA, NASA, CSIRO, QDAFF, IMOS, DERM
	Catch rate	SE Queensland	SST, Chl a., river discharge, eddy currents	NOAA, NASA, CSIRO, QDAFF, IMOS, DERM
Spanish mackerel	YCS	East coast	SST, Chl. a, river flow, SOI	Qld LTMP; CSIRO, BoM, DSITIA
	Catch rate	East coast	SST, Chl. a, river flow, SOI	Qld LTMP; CSIRO, BoM, DSITIA

*YCS = year Class Strength; NT = Northern Territory; GoC = Gulf of Carpentaria; EC = east coast; SST = Sea Surface Temperature; Chl.a = Chlorophyll a; LTMP = Long Term Monitoring Program; DSITIA = Department of Science, Information Technology, Innovation & the Arts; DERM = Department of Environment & Resource Management; ELF = Effects of Line Fishing Project; EAC = East Australian Current.

7.9   Vulnerability assessment

7.9.1   Assessment indicators and criteria

We developed a semi-quantitative approach to be used for the vulnerability assessments that used indicators for each of the elements Exposure, Sensitivity and Adaptive Capacity (Johnson and Welch 2010; Welch and Johnson 2013). *Exposure* indicators were developed based on the specific environmental variables predicted to be important for northern Australian fishery species and the criteria for these were developed to reflect the

environment the particular species lives in; for example, whether they were predominantly an estuarine or pelagic species. For each future scenario (e.g. 2030 A1FI, 2070 A1B, etc.) the exposure indicators were specific to the model projections that corresponded to that particular scenario. The indicators used for exposure for 2030 (A1FI & A1B), and their criteria, are shown in Table 7.11. Exposure indicators used for alternate future climate scenarios are provided based on those presented in Table 8.12 (Section 8.2).

The indicators and their criteria for **Sensitivity** were adapted from those developed by Pecl et al (2011a) who provide a detailed explanation of the development of these criteria. The indicators are based on different aspects of a species life history that can be affected by climate change: abundance, distribution and phenology. 'Abundance' relates to the capacity of a population to recover, which is essentially their productivity level. More productive species are deemed to be less sensitive to impacts because of their greater capacity to recover. 'Distribution' relates to the likelihood and capacity for a species to alter its range in response to environmental changes. 'Phenology' relates to the likelihood that environmental changes will result in changes to the timing of life cycle events (e.g. spawning). The Sensitivity indicators and their criteria are shown in Table 7.12.

The indicators for **Adaptive Capacity** were developed based on previous assessments and research (Allison et al. 2009; Johnson and Welch 2010; Marshall and Marshall 2007; Marshall et al. 2007; Pecl et al 2011a; Welch and Johnson 2013). Adaptive capacity can fall in two categories: the ability of the species to cope with changes (ecological), or the ability of participants in the industry (fishery) to cope with changes (socio-economic). We developed indicators for each of these categories, however, we only used the ecological Adaptive Capacity indicators when we applied our assessments, making these assessments ecologically-based only. We acknowledge that to truly assess the vulnerability of *fisheries* (as opposed to fishery species), the adaptive capacity of fishers and other industry members needs to be considered in the assessment process and to do this requires a dedicated consultation process, e.g. using surveys. However, it was not possible during this project to comprehensively consult with industry members in scoring the socio-economic indicators. The ecological and socio-economic indicators for Adaptive Capacity are shown in Table 7.13.

7.9.2 Assessment scoring

For each indicator, scores were assigned using Low (1), Medium (2) or High (3) and based on specified criteria (Tables 6.6 – 6.8). Pecl et al (2011a) demonstrated that this simple 3-level approach is sufficient for resolving species rankings, and for use by expert judgement while avoiding the need to determine precise rankings. For each element (e.g. Exposure) an index was calculated by dividing the total score by the number of indicators (i.e. the average score). The Potential Impact index was determined as the product of the Exposure and Sensitivity Indices (PI = E * S). Since vulnerability is defined as the *inability* to cope with

changes, the potential impact measured by the framework assumes a *negative* direction, however, some consequences of high exposure and high sensitivity are positive. For example, mud crab in north-western Australia have a high exposure to changes, largely due to their shallow water estuarine/nearshore habitat requirements, as well as relatively high sensitivity. However, the consequences of being exposed to increases in rainfall (and riverflow) and higher sea surface temperatures are likely to result in enhanced recruitment and catchability, as well as higher growth rates. To capture this we incorporated a 'Direction of impact' component with the Potential Impact score:

- Negative consequence = +1.0
- Neutral or unknown effect = 0.0
- Positive consequence = -1.0

The overall effect of adding this step in the scoring process moderated the level of vulnerability given to a species where the impact of climate change was likely to be positive. Therefore, for mud crab in north-western Australia where the consequence was actually positive, we subtracted 1.0 from the Potential Impact.

Since Adaptive Capacity (AC) is the inverse of both Exposure and Sensitivity, the final AC Index was determined based on the following process. First, the AC score was calculated as the average of the respective (ecological) indicator scores. These scores were then standardised to 1.00 with the highest average AC score given 1.00 and all other scores expressed as a proportion of this. That is, Standardised AC = Average AC/Maximum AC. The inverse was ten taken to derive the AC index. That is, AC index = 1 - Standardised AC. The vulnerability index was then calculated by the following: Vulnerability = (Potential Impact x AC index) + 1.

7.9.3 Vulnerability assessment process

The vulnerability assessments were done in a workshop setting with all project team members in attendance as well as other relevant experts (eg. a WA Fisheries representative). The project team, which comprised of scientists, managers, commercial and recreational fishers, with the addition of some key individuals, contained sufficient expertise and experience with the relevant species to provide comprehensive and informed assessments. A full list of workshop participants and their affiliations are given in Appendix 4. The assessment framework was explained to participants and a worked example was provided for discussion and clarification of the process, including making any minor refinements and/or additions to the framework.

Vulnerability assessments were then carried out for each individual species in the order of priority for each of the key regions as determined above using three major lines of evidence: (i) information summarised from the species reviews, (ii) information derived from project data analyses, and (iii) expert opinion. Scores were decided based on consensus among

workshop participants and, if necessary, the most conservative score was accepted for that indicator (i.e. for Exposure and Sensitivity the higher of the two possible scores was taken; for Adaptive Capacity the lower of the two possible scores was taken).

Table 7.11 Exposure indicators and their criteria. The indicators shown are based on changes in the respective variables projected for 2030. High (A1FI) and low (A1B) emission scenarios are similar for 2030.

Projections for 2030 (A1B & A1FI)		Low = 1	Medium = 2	High = 3
EXPOSURE	**SST increase 0.3 to 0.6 °C (EC, GoC); 0.6 to 0.9 °C (NWA)**	Adult spends <50% of time in surface (<25 m) waters	Adult spends 50-80% of time in surface (<25 m) waters	Adult spends 80-100% of time in surface (<25 m) waters
	Rainfall -10 to 0% (EC); 0 to +5% (NWA, GoC)	Spends no time in estuarine or freshwater habitats during any life history phase	Spends <50% of time in estuarine or freshwater habitats; no critical (larvae, juvenile, spawning) life history phase in these habitats	Spends >50% of time or has critical (larvae, juvenile, spawning) part of life cycle in estuarine or freshwater habitats
	pH decline 0.1 unit	Open ocean or deep water species	Continental shelf species	Inshore or estuarine species
	Salinity decline 0.1 psu	Open ocean or deep water species	Continental shelf species	Inshore or estuarine species
	Habitat changes (loss of productivity, structure or function) (nb. this incorporates sea level rise)	Species with wide habitat preferences	Species dependent on pelagic or mangrove/estuarine habitats	Species dependent on seagrass or coral reef habitats
	Altered large-scale currents: Stronger EAC; weaker Leeuwin current; GoC unknown	Live young/egg bearers or no dependence on large-scale wind/current for larval dispersal/settlement	Proximate dispersal/settlement of young not entirely dependent on large-scale wind/current dispersal	Dispersal/settlement of young 100% dependent on large-scale wind/currents
	More intense cyclones/storms (EC possibly fewer; NWA possibly more)	Deep water or highly mobile species	Shallow water (< 25 m) and moderately mobile species	Shallow water (< 25 m) or low mobility species
	Altered riverflow/nutrient supply: Reduction (EC) and potential increase linked to rainfall (NWA, GoC)	Spends no time in estuarine or freshwater habitats during any life history phase	Spends <50% of time in estuarine or freshwater habitats; no critical (larvae, juvenile, spawning) life history phase in these habitats	Spends >50% of time or has critical (larvae, juvenile, spawning) part of life cycle in estuarine or freshwater habitats

Table 7.12 Sensitivity indicators and their criteria (adapted from Pecl et al 2011a). Indicators are grouped into three categories of how the organism may be affected: abundance, distribution and phenology.

			Low = 1	Medium = 2	High = 3
Sensitivity	Abundance	Fecundity – egg production	>20,000 eggs/year	100-20,000 eggs/year	<100 eggs/year or live young
		Average age at maturity (females)	≤2 years	3-10 years	>10 years
		Generalist v. specialist (food & habitat)	Reliance on *neither* habitat or prey for any life history stage	Reliance on *either* habitat or prey for any life history stage	Reliance on *both* habitat and prey for any life history stage
	Distribution	Early development duration (dispersal capacity of larvae/young)	>8 weeks	2-8 weeks	<2 weeks or no larval stage
		Physiological tolerance of stock	Threshold unlikely to be exceeded for any climate variable	Physiological thresholds may be exceeded	Threshold likely to be exceeded for one or more climate variable
		Capacity for larvae to disperse	<100 km	100-500 km	>500 km
	Phenology	Reliance on environmental drivers (for spawning, breeding or settlement)	No apparent correlation to environmental variable	Weak correlation to environmental variable	Strong correlation to environmental variable
		Potential for timing mismatch of life cycle events (duration of spawning, moulting or breeding)	Continuous duration; >4 months	Moderate duration; 2-4 months	Brief duration; <2 months

Table 7.13 Adaptive capacity indicators and their criteria, grouped into ecological and socio-economic. NB. Adaptive capacity has the inverse effect compared to Exposure and Sensitivity. That is, low Sensitivity is a positive trait while low Adaptive Capacity is a negative trait.

			Low = 1	Medium = 2	High = 3
Adaptive Capacity	**Ecological**	**Stock status**	Overfished or on the verge of overfishing	Undefined	Sustainably fished
		Replenishment potential	Late maturing (>6 years), slow growth or few young	Matures at 3-6 years, moderate growth or moderate numbers of young	Early maturing, fast growth or many young
		Suitable alternate habitat availability	Low availability of habitat outside range *or* currently near northern edge of range	Some availability of habitat outside range *or* currently near middle of range	High availability of habitat outside range *and* currently near middle of range
		Species mobility	Low mobility; can travel <2 km/day	Moderately mobile; can travel 2-10 km/day	Highly mobile; can travel >10 km/day
		Non-fishing pressures on stock	Multiple chronic pressures (eg. poor water quality, disease, incidental catch)	Some acute pressures (eg. cyclones, storms, floods)	No or minimal other pressures
	Socio-economic	**Resource dependence**	No alternate species and/or significant gear/practice modifications required to target other species	Some alternate species that could be targeted with minor gear/practice modifications	Multiple alternate target species that could be targeted without any gear/practice modifications
		Ability to change fishing practices	Not able to change	Able to change with support	Able to change without support
		Climate change awareness	Unaware	Aware and no planning steps taken	Aware and has taken preparatory action
		Governance	Inflexible or non-existent	Flexible *or* adaptive (not both)	Flexible *and* adaptive

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7.9.4 Prioritising species for future action

The vulnerability assessment provides a robust basis for identifying species of highest concern and therefore priority species and fisheries for future action and/or further investigation, particularly for climate change adaptation. Relative vulnerability however should not be the only consideration for prioritisation of species. The relative 'fishery importance' of individual species should also be taken into account. To further assist managers, scientists and other fishery end-users in taking the next steps we incorporated the fishery importance 'scores' of each species derived through stakeholder consultation at the start of the project (see Tables 8.1-8.3) with relative vulnerability to prioritise species for future action. That is, species with higher vulnerability scores and higher fishery importance scores get higher priority.

7.10 Identifying adaptation options

To identify climate change adaptation options that were relevant and appropriate for particular fisheries and regions of northern Australian, two regional stakeholder workshops were conducted. The workshops were conducted in Darwin and Townsville and involved invited stakeholders that comprised of local fisheries and conservation managers, scientists, commercial fishers, recreational fishers, charter fishers, and fishing industry representatives (see Appendix 5 for lists of workshop participants and the workshop agenda).

At each of these workshops key project members presented the project, the vulnerability assessment framework, and the outputs of the preliminary assessments for key species relevant to the workshop region. To elicit adaptation options from workshop participants, outputs from the vulnerability assessments and species reviews were used to generate a summary of the likely impacts on key fishery species. The workshop used a series of breakout group sessions for discussing and identifying potential adaptation options and barriers to each of these impacts. The workshops were conducted to facilitate stakeholders to, as much as possible, be the ones who identified the key challenges and future considerations for their respective fisheries of interest. During each workshop, we also used the opportunity to obtain stakeholder perceptions on changes they have observed.

8   RESULTS/DISCUSSION

8.1   Species identification & prioritisation

There were a total of 40 species identified for the east coast, 36 species for the Gulf of Carpentaria, and 37 species for north-western Australia. Given the overlap in species across regions this was a total of 47 species for northern Australia. The identification of species, and their prioritisation, was not intended to provide an absolute species list for northern Australia. The list was intended to identify the key fishery species for the respective regions and to rank them as a guide *for this project* in carrying out the climate change vulnerability

assessments to ensure that the most important fishery species were included. Indeed, many fishery stakeholders across northern Australia had input to the whole process.

Not surprisingly given their widespread distribution, mud crab and barramundi were in the top 3 ranked species in all three regions. Other species ranked highly were banana and tiger prawns (EC and GoC), coral trout (EC), golden snapper and black jewfish (NWA), and king threadfin (GoC and NWA). The full species lists and their ranking order are provided for the east coast, the Gulf of Carpentaria, and north-western Australia in Tables 8.1, 8.2 and 8.3 respectively. Based on the fishery and ecological importance criteria for each species, which included social/cultural importance, economic importance, catch and ecological importance (see Table 7.1), the species with the five highest ranked scores from each region are shown in Tables 8.4, 8.5 and 8.6 for the east coast, the Gulf of Carpentaria and north-western Australia respectively.

Table 8.1 Species list and final rankings for fishery species identified for the east coast **based on total scores derived from scores for 'Fishery Importance' criteria (FI), and 'Climate change sensitivity' criteria (CC).**

Common name	Species name	FI	CC	Score
Coral trout	*Plectropomus* spp.	11.67	6.13	**17.79**
Mud crab	*Scylla serrata*	11.00	6.00	**17.00**
Barramundi	*Lates calcarifer*	11.00	5.94	**16.94**
Banana prawn	*Penaeus merguiensis*	10.67	6.25	**16.92**
Eastern king prawn	*Melicertus plebejus*	10.80	6.00	**16.80**
Tropical lobster	*Panulirus ornatus*	9.33	7.25	**16.58**
Tiger prawn	*Penaeus esculentus/P. semisulcatus*	10.40	6.17	**16.57**
Spanish mackerel	*Scomberomorus commerson*	11.17	5.33	**16.50**
Red spot king prawn	*Penaeus longistylus*	10.00	6.17	**16.17**
Blacktip sharks	*Carcharhinus limbatus/C. tilstoni*	9.17	6.50	**15.67**
Red throat emperor	*Lethrinus miniatus*	9.00	6.38	**15.38**
Spot tail shark	*Carcharhinus sorrah*	8.75	6.50	**15.25**
Billfish (Black marlin)	*Istiompax indica*	10.00	5.13	**15.13**
Moreton Bay bug	*Thenus orientalis*	9.17	5.92	**15.08**
Red emperor	*Lutjanus sebae*	9.17	5.92	**15.08**
Grey mackerel	*Scomberomorus semifasciatus*	9.00	5.92	**14.92**
Sand fish	*Holothuria scabra*	7.50	7.38	**14.88**
Pigeye shark	*Carcharhinus*	8.25	6.63	**14.88**
Saucer scallop	*Amusium japonicum*	8.83	5.92	**14.75**
Spotted mackerel	*Scomberomorus munroi*	8.67	5.69	**14.36**
King threadfin	*Polydactylus macrochir*	8.00	6.25	**14.25**
Spanner crab	*Ranina ranina*	8.17	5.92	**14.08**
Blue threadfin	*Eleutheronema tetradactylum*	7.83	6.08	**13.92**
Scalloped hammerhead	*Spyhrna lewini*	7.25	6.63	**13.88**
Whiting	*Sillago ciliata*	8.00	5.50	**13.50**
Barred javelin	*Pomadasys kaakan*	7.67	5.78	**13.44**
Mangrove jack	*Lutjanus argentimaculatus*	7.33	5.67	**13.00**
Pikey bream	*Acanthopagrus berda*	7.17	5.83	**13.00**
Golden snapper	*Lutjanus johnii*	7.17	5.75	**12.92**
Dusky flathead	*Platycephalus fuscus*	7.17	5.67	**12.83**
Crimson snapper	*Lutjanus erythropterus*	6.60	5.75	**12.35**
Saddle tail snapper	*Lutjanus malabaricus*	6.60	5.75	**12.35**
School mackerel	*Scomberomorus*	6.00	6.25	**12.25**
Black jewfish	*Protonibea diacanthus*	6.33	5.83	**12.17**
Grass emperor	*Lethrinus laticaudis*	6.17	5.92	**12.08**
Spangled emperor	*Lethrinus nebulosus*	5.83	5.92	**11.75**
Goldband snapper	*Pristipomoides*	5.75	5.63	**11.38**
Black spot cod	*Epinephelus malabaricus*	5.00	5.88	**10.88**
Gold spot cod	*Epinephelus coiodes*	5.00	5.88	**10.88**
Blue swimmer crab	*Portunus pelagicus*	5.00	5.00	**10.00**

Table 8.2 Species list and final rankings for fishery species identified for the Gulf of Carpentaria **based on total scores derived from scores for 'Fishery Importance' criteria (FI), and 'Climate change sensitivity' criteria (CC).**

Common name	Species name	FI	CC	Score
Banana prawn	*Penaeus merguiensis*	10.67	6.00	**16.67**
Mud crab	*Scylla serrata*	10.75	5.75	**16.50**
Barramundi	*Lates calcarifer*	10.75	5.67	**16.42**
Tiger prawn	*Penaeus esculentus/P. semisulcatus*	10.33	5.92	**16.25**
King threadfin	*Polydactylus macrochir*	10.00	6.13	**16.13**
Grey mackerel	*Scomberomorus semifasciatus*	10.00	5.38	**15.38**
Tropical lobster	*Panulirus ornatus*	8.00	6.63	**14.63**
Billfish (Sailfish)	*Istiophorus platypterus*	8.00	6.38	**14.38**
Blacktip sharks	*Carcharhinus limbatus/C. tilstoni*	9.50	4.88	**14.38**
Spanish mackerel	*Scomberomorus commerson*	9.25	4.75	**14.00**
Golden snapper	*Lutjanus johnii*	7.00	6.50	**13.50**
Scalloped hammerhead	*Sphyrna lewini*	8.00	5.38	**13.38**
Blue threadfin	*Eleutheronema tetradactylum*	7.75	5.50	**13.25**
Mullet	*Liza vaigiensis*	8.00	5.25	**13.25**
Spot tail shark	*Carcharhinus sorrah*	8.33	4.88	**13.21**
Pigeye shark	*Carcharhinus amboinensis*	7.50	5.38	**12.88**
Red emperor	*Lutjanus sebae*	6.33	6.38	**12.71**
Sand fish	*Holothuria scabra*	5.67	7.00	**12.67**
Mangrove jack	*Lutjanus argentimaculatus*	6.25	6.38	**12.63**
Black jewfish	*Protonibea diacanthus*	6.25	6.00	**12.25**
Garfish	*Hyporamphus* spp.	7.00	5.25	**12.25**
Grass emperor	*Lethrinus laticaudis*	5.67	6.38	**12.04**
Barred javelin	*Pomadasys kaakan*	7.75	4.00	**11.75**
Sardines/herring	Family Clupeidae	6.00	5.75	**11.75**
Coral trout	*Plectropomus* spp.	5.33	6.13	**11.46**
Spangled emperor	*Lethrinus nebulosus*	5.00	6.38	**11.38**
Bugs	*Thenus orientalis*	5.67	5.50	**11.17**
Crimson snapper	*Lutjanus erythropterus*	5.00	5.88	**10.88**
Saddle tail snapper	*Lutjanus malabaricus*	5.00	5.88	**10.88**
Saucer scallops	*Amusium japonicum*	5.00	5.75	**10.75**
Goldband snapper	*Pristipomoides multidens*	4.67	5.75	**10.42**
Pikey bream	*Acanthopagrus berda*	6.00	4.00	**10.00**
Dusky flathead	*Platycephalus fuscus*	4.50	4.75	**9.25**
Spotted mackerel	*Scomberomorus munroi*	4.67	4.50	**9.17**
Longtail tuna	*Thunnus tonggol*	5.00	4.00	**9.00**
Whiting	*Sillago ciliata*	5.00	3.88	**8.88**

Table 8.3 Species list and final rankings for fishery species identified for north-western Australia **based on total scores derived from scores for 'Fishery Importance' criteria (FI), and 'Climate change sensitivity' criteria (CC).**

Common name	Species name	FI	CC	Score
Barramundi	*Lates calcarifer*	11.50	5.63	**17.13**
Mud crab	*Scylla serrata*	11.00	6.13	**17.13**
Black jewfish	*Protonibea diacanthus*	10.50	6.13	**16.63**
Golden snapper	*Lutjanus johnii*	10.00	6.63	**16.63**
King threadfin	*Polydactylus macrochir*	10.00	6.00	**16.00**
Spanish mackerel	*Scomberomorus commerson*	11.00	4.88	**15.88**
Sand fish	*Holothuria scabra*	8.50	7.13	**15.63**
Billfish (Sailfish)	*Istiophorus platypterus*	8.00	7.50	**15.50**
Grey mackerel	*Scomberomorus semifasciatus*	9.50	5.50	**15.00**
Scalloped hammerhead	*Sphyrna lewini*	9.50	5.50	**15.00**
Red emperor	*Lutjanus sebae*	8.00	6.50	**14.50**
Goldband snapper	*Pristipomoides multidens*	8.50	5.88	**14.38**
Blacktip sharks	*Carcharhinus limbatus/C. tilstoni*	9.00	5.00	**14.00**
Pigeye shark	*Carcharhinus amboinensis*	8.50	5.50	**14.00**
Spot tail shark	*Carcharhinus sorrah*	9.00	5.00	**14.00**
Crimson snapper	*Lutjanus erythropterus*	7.50	6.00	**13.50**
Mangrove jack	*Lutjanus argentimaculatus*	7.00	6.50	**13.50**
Saddle tail snapper	*Lutjanus malabaricus*	7.50	6.00	**13.50**
Mullet	*Liza vaigiensis*	8.00	5.25	**13.25**
Blue threadfin	*Eleutheronema tetradactylum*	7.50	5.63	**13.13**
Coral trout	*Plectropomus* spp.	6.50	6.25	**12.75**
Tropical lobster	*Panulirus ornatus*	6.00	6.75	**12.75**
Grass emperor	*Lethrinus laticaudis*	6.00	6.38	**12.38**
Barred javelin	*Pomadasys kaakan*	8.00	4.25	**12.25**
Garfish	*Hyporamphus* spp.	7.00	5.25	**12.25**
Sardines/Herrings	Family Clupeidae	6.00	5.75	**11.75**
Bugs	*Thenus orientalis*	5.00	6.25	**11.25**
Saucer scallops	*Amusium japonicum*	5.00	6.25	**11.25**
Banana prawn	*Penaeus merguiensis*	5.00	6.00	**11.00**
Spangled emperor	*Lethrinus nebulosus*	4.50	6.50	**11.00**
Tiger prawn	*Penaeus esculentus/P. semisulcatus*	5.00	6.00	**11.00**
Giant trevally	*Caranx ignobilis*	6.00	4.75	**10.75**
Golden trevally	*Gnathodon*	5.00	5.00	**10.00**
Spotted mackerel	*Scomberomorus munroi*	5.00	4.63	**9.63**
Dusky flathead	*Platycephalus fuscus*	4.00	5.25	**9.25**
Pikey bream	*Acanthopagrus berda*	5.00	4.25	**9.25**
Whiting	*Sillago ciliata*	4.50	4.13	**8.63**

Table 8.4 Fishery species with the five highest ranked scores for the east coast based only on the fishery/ecological importance attributes.

Common name	Species name	FI
Coral trout	*Plectropomus* spp.	11.67
Spanish mackerel	*Scomberomorus commerson*	11.17
Mud crab	*Scylla serrata*	11.00
Barramundi	*Lates calcarifer*	11.00
Eastern king prawn	*Melicertus plebejus*	10.80
Banana prawn	*Penaeus merguiensis*	10.67

Table 8.5 Fishery species with the five highest ranked scores for the Gulf of Carpentaria based only on the fishery/ecological importance attributes.

Common name	Species name	FI
Mud crab	*Scylla serrata*	10.75
Barramundi	*Lates calcarifer*	10.75
Banana prawn	*Penaeus merguiensis*	10.67
Tiger prawn	*Penaeus esculentus/semisulcatus*	10.33
King threadfin	*Polydactylus macrochir*	10.00
Grey mackerel	*Scomberomorus semifasciatus*	10.00
Blacktip sharks	*Carcharhinus limbatus/tilstoni*	9.50

Table 8.6 Fishery species with the five highest ranked scores for north-western Australia based only on the fishery/ecological importance attributes.

Common name	Species name	FI
Barramundi	*Lates calcarifer*	11.50
Mud crab	*Scylla serrata*	11.00
Spanish mackerel	*Scomberomorus commerson*	11.00
Black jewfish	*Protonibea diacanthus*	10.50
Golden snapper	*Lutjanus johnii*	10.00
King threadfin	*Polydactylus macrochir*	10.00
Grey mackerel	*Scomberomorus semifasciatus*	9.50
Scalloped hammerhead	*Sphyrna lewini*	9.50

8.2   Observed and projected climate for northern Australia

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8.2.1   Northern Australia's observed climate and recent trends

The natural ecosystems that northern Australian fisheries rely on have evolved to operate within a specific range of prevailing local climatic conditions – the coping range (Jones and Mearns 2005). Any changes in these specific climate conditions will influence fisheries resources – stocks, species, populations and communities – and the habitats that support them. Therefore understanding the observed climate is important when seeking to assess how these resources are likely to respond under future climate change. Known sensitivities provide insight into potential impacts and ultimately the sustainability of fisheries over this century. The information aims to provide a picture of the observed ocean climate that fisheries species (prioritised for this study) have experienced, focusing on the variables that they are most likely to be sensitive to, and future projections for these variables. The projections focus on the three project regions: the East Coast (EC), Gulf of Carpentaria (GoC) and northwest Australia (NWA). Table 8.7 summarises the variables, expected fisheries sensitivities, level of confidence in the data and data sources.

Table 8.7 Data considerations for different climate variables

Variable	Expected fisheries sensitivity	Level of confidence	Source	Spatial coverage
SST	Low – Very high	High	CSIRO/BoM/NOAA	EC; GoC; NWA
Ocean temp at >10 m	Low – Very high	Low	BoM/UCSD	EC; GoC; NWA
Rainfall	Low – High	High	BoM/QDNRM	EC; GoC; NWA
Riverflow/nutrient supply	Low – High	Mod – High	QDNRM	By catchment
Ocean pH	Moderate*	High	CSIRO/NOAA/NCAR	EC; GoC; NWA
Storms & cyclones	Moderate*	High	CSIRO/BoM	By region
Sea level	Low	High	BoM/NASA	EC; GoC; NWA
Ocean circulation	Moderate*	Low	CSIRO	EC; GoC; NWA

The information was drawn from a range of sources, particularly climate modelling for Australia (CSIRO and BoM 2007, Poloczanska et al. 2012), as well as relevant key literature (e.g. BoM and CSIRO 2011, Church and White 2011, Lough and Hobday 2011). Observed

climate data was used to conduct the climate-response analyses for priority fisheries species, with an example of the variables that are likely to be prioritised for analyses for different fisheries provided in Table 7.5 (and see Appendix 6 for all species examined). These results will provide an indication of how fisheries species are likely to respond to future climate projections and how this might influence fisheries activities and management.

8.2.2   Observed climate trends

Average seasonal surface climate in northern Australia is dominated by large-scale global circulation systems, such as the south-easterly trade winds and the Australian summer monsoon westerly circulation. These effectively divide the year into a warm summer wet season (October to March) and a cooler winter dry season (April to September). Tropical cyclones are also an important feature of the summer monsoon and occur mainly between November and May (Lough 2007).

Sea surface temperatures
Monthly mean air and sea surface temperatures (SST) in northern Australia show a similar spatial distribution, however SST varies more slowly than air temperatures due to the heat capacity of water. As such, SST lag air temperatures on a seasonal timescale by about 1 month. The annual maxima are usually observed in February/March and the minima are usually observed in August/September. Greatest seasonal warming of SST occurs from September to October (+1.4 to 1.7°C) and greatest seasonal cooling from May to June (-1.1 to 1.8°C). SST tends to be warmer than air temperatures throughout the year, the difference being greater in winter than in summer.

On the tropical east coast, monthly mean SST range from 26 – 30.5 °C (from south to north) in the summer monsoon and 21 – 27 °C (from south to north) in the winter dry season. In the Gulf of Carpentaria SST range from 30 – 32 °C in the summer and are around 26 °C in winter. In northern WA the range is 29.4 – 30.5 °C (from south to north) in the summer monsoon and 24.8 – 26.6 °C (from south to north) in the winter dry season (Table 8.8). SST in NWA is also influenced by the warm Indonesian Throughflow from the western Pacific and northern Australian region, which eventually forms the Leeuwin Current along the WA coast.

The annual range of SST is approximately 4°C in the northern tropics and approximately 6°C in the southern tropics. However, these data are based on large-scale averages and the range of SST variability observed at a particular site can be much greater. For example, at the offshore Myrmidon Reef in the Great Barrier Reef (GBR), the average diurnal SST range is 1°C and average daily SST vary between a minimum of 24°C in late August to a maximum of 29°C in early February (4.8°C range; Lough and Hobday 2011). These large-scale averages

also disguise the tendency for SST in nearshore shallow waters to be warmer in summer and cooler in winter compared to offshore deeper waters (Lough 2007).

Table 8.8 Average SST, maxima and minima (^{o}C) for representative stations within the three project regions (Source: Bureau of Meteorology).

Region	SST mean (°C)	SST maximum (°C)	SST minimum (°C)
East coast: GBR (Cairns[1])	26.3	29 (28.7)	22 (24)
Gulf of Carpentaria: Groote Eylandt (Weipa[2])	29.4 (28.3)	31.8 (30)	26.2 (26.3)
Northwest Australia: Melville Island (Broome[1])	28.4 (26.8)	30.1 (29)	26.4 (23.4)

Significant warming is already evident in Australia's oceans (Lough 2009, Lough et al. 2012) with warming greatest off southeast Australia and in the Indian Ocean off southwest Australia, as well as substantial warming of the tropical Pacific Ocean over recent decades (BoM and CSIRO 2011). The recent warming of Australian waters has seen average SST increase above early 20th century records (1910–1929) by +0.68 °C. The rate of temperature rise in Australian waters has accelerated since the mid-20th century; from 0.08 °C per decade in 1910-2011 to 0.11 °C per decade from 1950-2011 (Lough et al. 2012)(Figure 8.1). The evidence for significant ocean warming both at the surface and through the water column is supported by global SST compilations, and continuous *in situ* coastal observations (e.g. Ridgway 2007, Caputi et al. 2009, Lough et al. 2010). This warming has been accompanied by increasing sea-surface salinity (Pearce and Feng 2007, Thompson et al. 2009) and changes in 'climate zones'. In Australia's coastal waters between 10.5° S and 29.5° S, warming has resulted in southward shifts of climate zones by ~200 km along the east coast and ~100 km along the west coast over the period 1950-2007 (Lough 2008).

Rainfall and river flow
Australia has a high degree of inter-annual and decadal rainfall variability, making understanding the significance of rainfall extremes and changes in averages difficult to detect (CSIRO and BoM 2007, Gallant et al. 2007, Hennessy et al. 2008). In the northern tropics, the summer monsoon circulation brings the majority of the annual rainfall with approximately 80% of the annual total occurring in the wet season. On the EC, rainfall typically occurs on only 30% of days in summer and only 14% of days in winter. The GoC is in

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[2] http://www.metoc.gov.au/products/data/aussst.php

the wet-dry tropics and experiences high rainfall, for example Weipa has a mean annual rainfall of 1768 mm, the majority of which falls between November and April. The Kimberley coast region of NWA has both arid and wet tropical environments with annual average rainfall varying between <200 mm and >1000 mm, respectively.
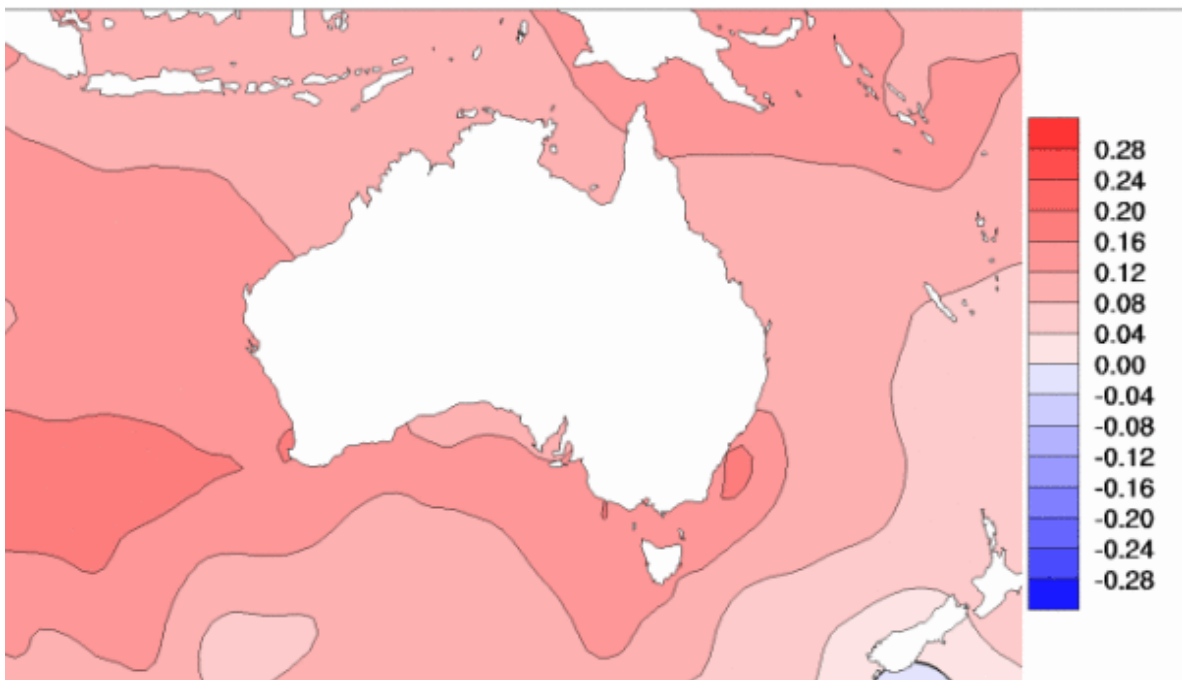
Figure 8.1 Trend in SST for the Australia region (°C/10yr) from 1950 – 2012 (Source: Bureau of Meteorology).

Due to the high spatial and temporal variability of rainfall in tropical Australia, the long-term average is not a good guide to the amount of rain that can be expected. The median is more appropriate as it is not influenced by extreme high and low values that are common. All coastal rainfall sites show maximum rainfall and greatest variability during the summer monsoon from about December to March (Table 8.9)(Lough and Hobday 2011).

The highly seasonal and variable rainfall regime of northern Australia also results in highly variable river flows. For example, the majority (~ 80%) of total river flow into EC coastal waters occurs between 17° S and 23° S with greatest annual flow in March, after the rainfall maxima. High rainfall variability results in Australian river flows being among the most variable in the world (Finlayson and McMahon 1988; Ward et al 2010). The larger rivers emptying into the GoC are the Wenlock, Archer, Holroyd, Mitchell, Staaten, and Gilbert, and again highly variable rainfall influences river flow, with greatest flows at the end of the wet season. The landscape of northwest WA has a comprehensive network of large river systems that have highly variable river flows that peak during the late wet season with tropical cyclones being a primary driver of significant rainfall and hence river flows (Lavender and Abbs 2013). The Pilbara region has the Ashburton, De Grey, Robe and Fortesque Rivers.

Major rivers in the Kimberley include the Ord River in the east and the Fitzroy River spans ~750 km and is said to be the largest river in Australia when in flood. The meeting of five river systems near Wyndham causes significant flows out to Cambridge Gulf during the wet season. These highly variable river flows regulate many processes in freshwater and coastal environments and shape ecosystem dynamics (e.g. Leigh et al. 2010, Puckridge et al. 2010).

Table 8.9 Average monthly rainfall (mm) for representative stations within the three project regions based on available records to date (e.g. 1941 for Broome, 1914 for Weipa). Bold cells show wettest months.

Region	J	F	M	A	M	J	J	A	S	O	N	D
East coast (Lockhart River[2])	**396**	**391**	**452**	297	107	58	44	29	16	29	72	208
East coast (Gladstone[2])	**153**	**146**	90	48	59	38	35	33	27	60	72	**132**
Gulf of Carpentaria: (Weipa[3])	**456**	**441**	**349**	109	16	4	2	3	6	27	103	265
Gulf of Carpentaria: (Nhulunbuy[2])	228	234	268	233	83	17	13	4	4	11	26	**194**
Northwest Australia (Broome[2])	**179**	**180**	**102**	26	27	20	7	2	1	1	9	56

Seasonal, inter-annual and longer-term rainfall variability across Australia is largely controlled by external factors, for example, El Niño-Southern Oscillation (ENSO) events have long been recognised as the primary source of inter-annual variability across much of the eastern part of the country (e.g. Troup 1965, Allan et al. 1996), although effects vary across seasons and region (Risbey et al. 2009). Over eastern Australia this high rainfall variability associated with ENSO also results in river discharge being highly sensitive to ENSO (Ward et al. 2010).

Documented trends of wetter summer conditions in northwest Australia (Shi et al. 2008, Smith et al. 2008) appear to be part of significant changes in large-scale atmospheric circulation patterns, including a more intense subtropical ridge along the east coast (Larsen and Nicholls 2009, Nicholls 2010). The east coast has experienced substantial rainfall declines since 1950, partly reflecting a very wet period around the 1950s, and recent years that have been unusually dry. However, this trend has changed in recent years, with the

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[3] http://www.bom.gov.au/jsp/ncc/cdio/weatherData

2010/11 and 2012/13 wet seasons having extreme rainfall and flood events that dominated the Queensland summer under a climate system that is warmer and moister (Climate Commission 2013). In contrast, northwest Australia has experienced a consistent increase in rainfall since the 1970s (CSIRO and BoM 2007)(Figure 8.2).
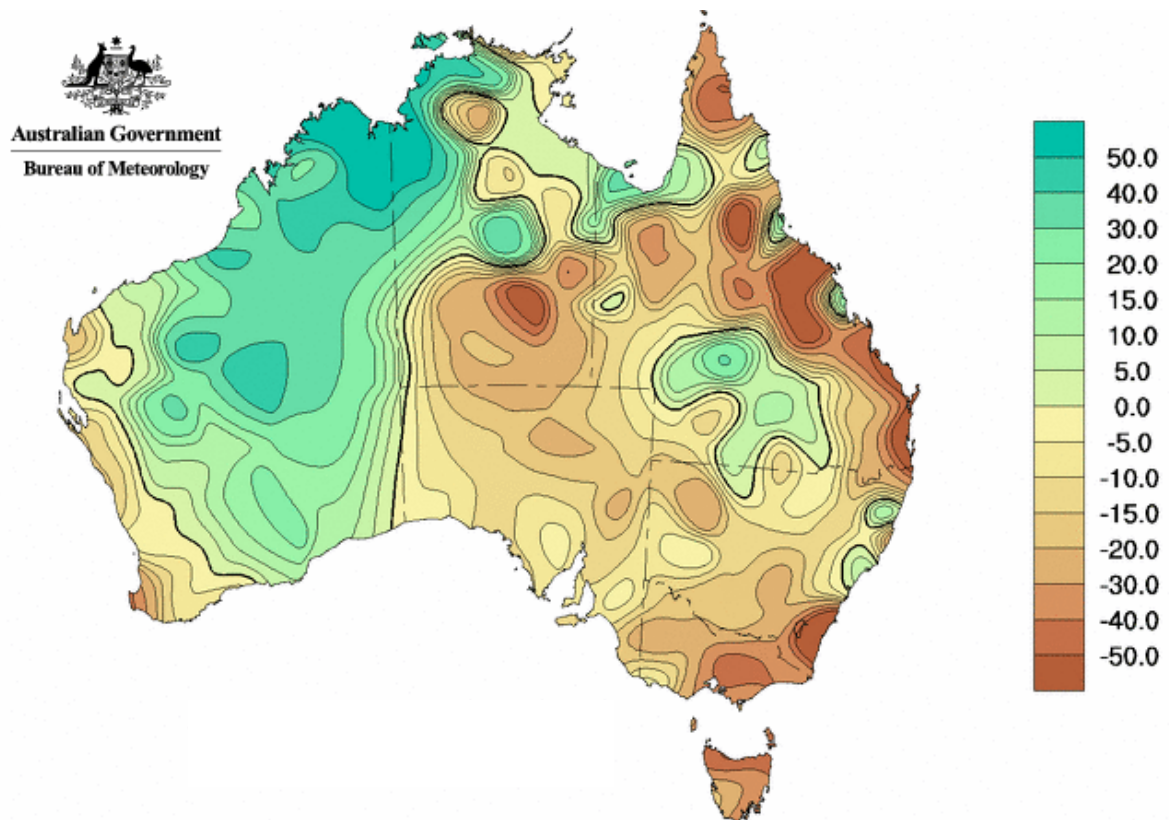
Figure 8.2 Trend in annual total rainfall 1970 – 2012 (mm/10yr) with green representing an increase in rainfall and brown a decrease over time (Source: Bureau of Meteorology).

Trends in extreme daily rainfall across Australia show that from 1970 to 2012, there have been increases in north-western and central Australia, but decreases along the EC. Trends in most extreme rainfall events are rising faster than trends in the mean. Since the start of the 20th century, the period with the lowest rainfall was from the 1930s to the early 1940s. In addition, recent Australian droughts have been accompanied by higher surface temperatures due to global warming, which may have exacerbated the impact of drought in regions where warming increases water demand and surface evaporation (Lough and Hobday 2011).

Although clear evidence is now emerging for a recent acceleration in the global hydrological cycle (Helm et al. 2010), assessing the magnitude and significance of observed rainfall and river flow changes across Australia is hindered by the high inter-annual rainfall variability (Lough and Hobday 2011).

Ocean chemistry

The oceans absorb carbon dioxide (CO2) from the atmosphere and are estimated to have absorbed about a third of the excess CO_2 released into the atmosphere by human activities in the past 200 years (Doney et al 2009). Global estimates of changes in ocean chemistry are used as very little is known about baseline conditions, and any variation in Australian waters is complex and variable both spatially and temporally (e.g. Gagliano et al. 2010). In addition, most measurements have been made for open-ocean waters, which are not representative of coastal or nearshore waters (e.g. McNeil 2010) and especially coral reefs, which modify their own water chemistry (Anthony et al. 2011).

Changes in water chemistry are the result of CO_2 absorption, with about half of this anthropogenic CO_2 remaining in the upper 10% of oceans (less than 1,000 m depth) due to slow ocean mixing processes. This absorbed CO_2 is resulting in chemical changes in the ocean, causing an estimated 0.1 decrease in oceanic pH since 1750, representing a 30% increase in hydrogen ion (acid) concentration (Ganachaud et al. 2011, Howard et al. 2012). Although this pH change is not uniform for all Australian waters, there is not high enough data resolution to provide spatially refined estimates.

Sea level

As global climate warms, sea level rises due to thermal expansion of the oceans and the contribution of additional water through the melting of glaciers and continental ice sheets. As a result, global sea level appears to be rising at a rate of 1 to 2 mm per year. A recent reconstruction of global mean sea level from 1870 indicates that between January 1870 and December 2004, global sea level rose by 195 mm (Figure 8.3). There is also observational evidence (matching climate model simulations) of a significant acceleration in the rate of global sea level rise of 0.13 ± 0.006 mm per year (Church and White 2006).

Globally, average sea level has risen 20 cm since the late 19th century, largely as a result of thermal expansion, with a relatively minor contribution, so far, from melting land ice (Bindoff et al. 2007). The average rate between 1950 and 2000 was 1.8 ±0.3 mm per year, but for the period when satellite data are available (i.e. from 1993), the rate increased to 3 mm per year. Since 1990, the observed rate of global sea level rise corresponds to the upper limit of IPCC projections (IPCC 2007). For the period 1993 to 2011, sea level rose at all of the Australian coastal sites monitored, with substantial variability in trends from location to location (Figure 8.4). Over the period 1920 to 2000 the estimated average relative sea level rise around Australia was 1.2 mm per year (Lough and Hobday 2011).
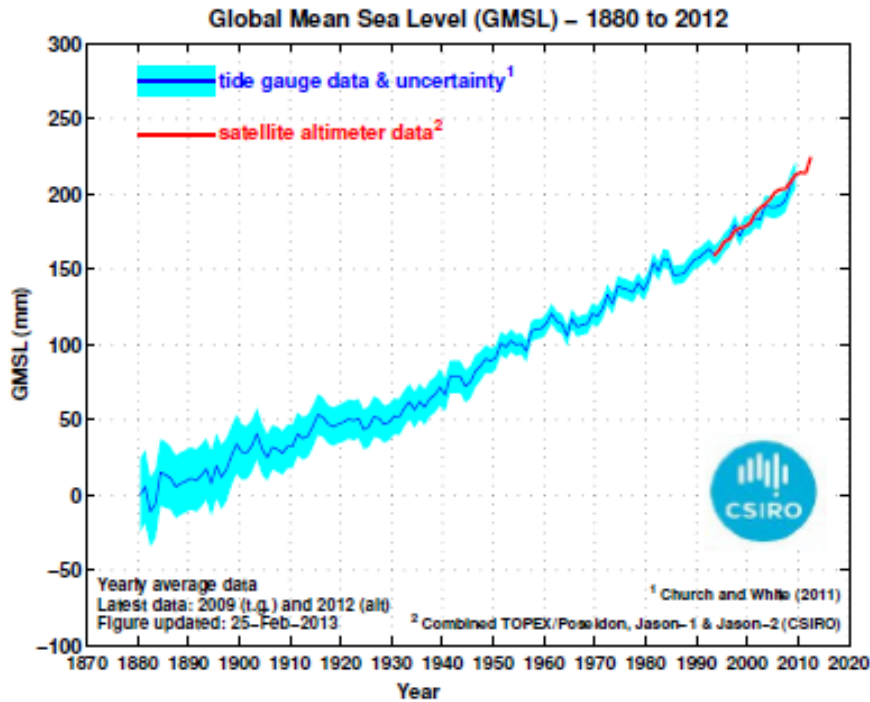
Figure 8.3 Global mean sea level 1880 – 2012 (Source: CSIRO, Church and White 2011).

Sea levels are rising around Australia, with fastest rates currently in northern Australia. Rising sea level also affects the frequency of high sea-level events (e.g. storm surge inundation) on annual to decadal timescales and these have increased by a factor of three during the 20th century (Church et al. 2012). This regional variation in the magnitude of sea-level rise is linked with inter-annual climate variability (e.g. ENSO), and changes in ocean circulation (e.g. increased southward penetration of the East Australian Current) and atmospheric circulation dynamics (Church et al. 2009). Sea-level rise is, therefore, not uniform in either the Indian or Pacific Oceans (Han et al. 2010).

Tropical cyclones

Tropical cyclones occur during the summer monsoon season and are destructive weather systems that affect tropical Australia (Figure 8.5). Conditions suitable for tropical cyclone development occur from November through May on the EC, GoC and NWA, with highest numbers usually experienced in January and February. In NWA the chance of experiencing an intense cyclone (category 4 or 5) is highest in March and April. The northwest WA coastline between Broome and Exmouth is the most cyclone-prone region of the Australian coastline, having the highest frequency of coastal crossings and the highest incidence of cyclones in the southern hemisphere (Lough 1998, BoM 2013). On average about five tropical cyclones occur during each tropical cyclone season over the warm ocean waters off the northwest coast between 105°E and 125°E (BoM 2013). Tropical cyclones occur most frequently between latitudes 16° S to 18° S and less often between latitudes 10° S to 12° S. Tropical cyclones bring destructive winds and waves and, when making landfall, can cause

elevated sea levels and destructive storm waves (storm surge) as well as heavy rainfall and rapid increases in river flows (Lough 2007).
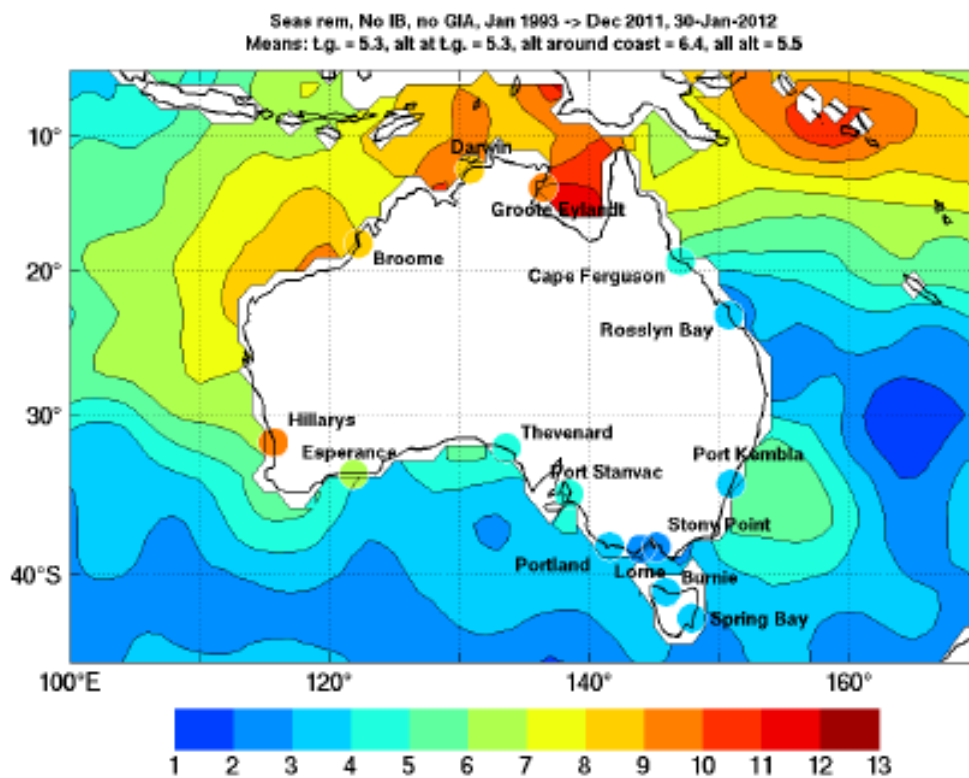
Figure 8.4 Australian sea-level trend (mm/yr) from 1993 – 2011 (Source: Church et al. 2012).
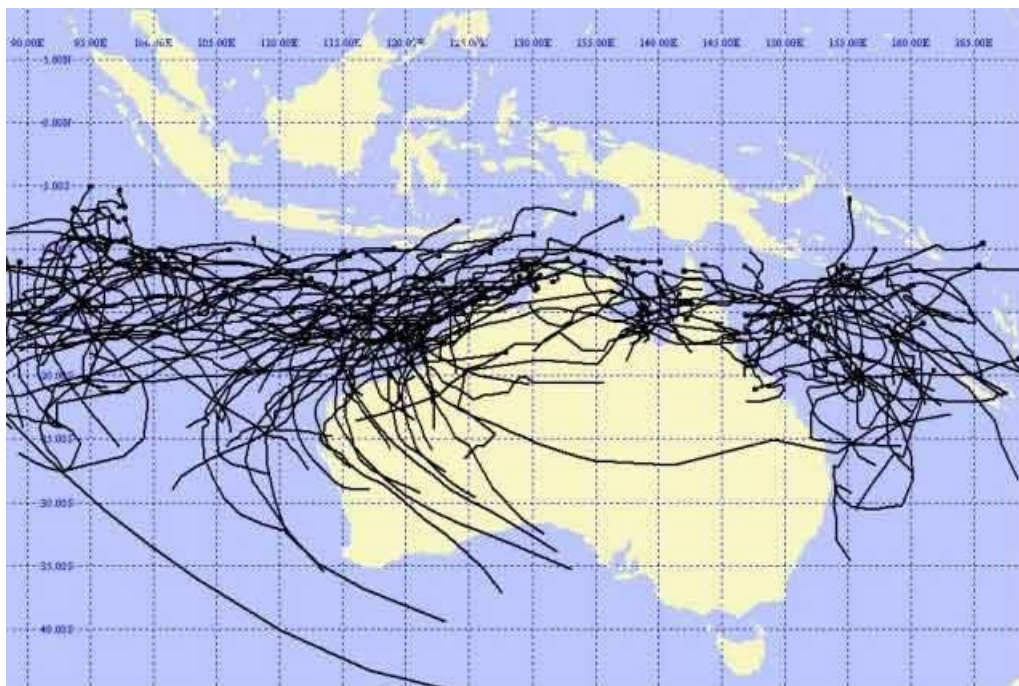
Figure 8.5 Tropical cyclone tracks in the Australian region from 1989/90 to 2002/03 (Source: Bureau of Meteorology).

Detecting trends in tropical cyclone frequency and intensity is difficult due to the high natural variability in their occurrence and that it is probably only since the advent of satellites that all tropical cyclones have been identified (Knutson et al. 2010). Nicholls et al. (1998) provided evidence of an apparent decline in the number of weak tropical cyclones and a slight increasing trend of more intense tropical cyclones in the Australian region over the period from 1969–1970 to 1995–1996 based on satellite observations. This trend has been attributed, in part, to improved distinction of tropical cyclones from other tropical storms. Hassim and Walsh (2008) compared eastern and western Australian tropical-cyclone regions from 1969–1970 to 2004–2005 and found evidence that the number, duration and maximum intensity of severe tropical cyclones off NWA have been increasing since the 1980s. However, on the EC, the number has decreased, with no obvious trend in either intensity or duration. There has been no observed change in the latitudinal distribution of tropical-cyclone activity. The GBR on the EC has however experienced six severe tropical cyclones between 2005 and 2011 (GBRMPA 2011).

El Niño – Southern Oscillation

A major source of inter-annual climate variability in northeast Australia (affecting the EC and GoC) is the El Niño – Southern Oscillation (ENSO) phenomenon (Lough and Hobday 2011). ENSO describes the aperiodic variations in the ocean-atmosphere climate of the tropical Pacific, which causes climate anomalies in many parts of the tropics and extra-tropics. ENSO has two phases:

      1) El Niño events when the eastern equatorial Pacific is unusually warm, and
      2) La Niña events when the eastern equatorial Pacific is unusually cold.

Events typically evolve over 12 to 18 months and, once initiated, their development is somewhat predictable with distinct climate anomalies occurring in northeast Australia. During typical El Niño events, the summer monsoon circulation is weaker than normal associated with higher sea level pressure and more south-easterly winds. Cloud cover is reduced increasing radiation, and rainfall and river flows are considerably lower than normal. In typical La Niña events, the summer monsoon circulation is stronger than normal with lower sea level pressure and more north-westerly winds. Cloud cover, rainfall and river flows are higher than average. SST anomalies are more marked during El Niño than La Niña events, with warmer than average SST occurring during the summer wet season. The differences in the strength of the summer monsoon circulation with ENSO also results in marked differences in the occurrence of tropical cyclones with much less activity during El Niño years. Overall, the level of disturbance is greater during La Niña events when the more vigorous summer monsoon circulation and heightened tropical cyclone activity causes enhanced rainfall and river flow. This can lead to reduced salinity and higher turbidity of nearshore waters and increased levels of physical disturbance. Suppression of the summer monsoon and tropical cyclone activity during El Niño events is associated with reduced rainfall and river flow and maintenance of more winter-like conditions (Lough 2007).

Instrumental and palaeo-climate records show large variations in the frequency and intensity of ENSO, and the impact of ENSO on Australia has varied from decade to decade. This is partly driven by the Pacific Decadal Oscillation (PDO, Mantua et al 1997; Power et al 1999). While there has been an apparent increase in the frequency of El Niño events in recent years, there is no consensus amongst global climate models that climate change should cause such an increase; the increase might therefore reflect natural variability. The relationship between the SOI and Australian temperatures and rainfall has changed. For example, Australia-wide rainfall and temperatures since the mid-1970s have been higher for any given value of the SOI than previously (CSIRO and BoM 2007).

Although El Niño or La Niña events show some common features, no two evolve in exactly the same way (Trenberth and Stepaniak 2001) and recently it has been suggested that ENSO events have shifted from those dominated by warming or cooling centred in the eastern equatorial Pacific to events characterised by warming or cooling in the central equatorial Pacific (Ashok et al. 2007). Whether this is a signal of global warming is not clear yet (Lough and Hobday 2011).

Wind and ocean currents
The prevailing wind conditions that influence the east coast of Australia are the southeast trade winds. The impact of the trade winds is greatest from April to September with winds of 45 to 55 km/h often observed north of Cooktown. Trade winds are at their strongest when a slow-moving high pressure system is located off the east coast of Australia in the Tasman Sea[4].

In the GoC, the prevailing winds are south-easterly during the dry season and north-westerly during the wet season. These trade winds are driven by the sub-tropical ridge; an extensive area of high pressure that lies across southern Australia in winter, and further south in summer. The trade winds tend to be strongest in winter when high-pressure systems are more intense (April to September), directing cool south-easterly winds towards northern Australia[5]. Mid-latitude westerly winds appear to have decreased in most seasons from 1979 to the late 1990s and there has been a 20% reduction in the strength of the subtropical jet over Australia (CSIRO and BoM 2007). In NWA the prevailing wind conditions are westerly/north-westerly during the summer and more variable in winter[6].

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[4] http://www.climatekelpie.com.au/understand-climate/weather-and-climate-drivers/queensland#TradeWinds

[5] http://climatekelpie.com.au/understand-climate/weather-and-climate-drivers/northern-territory#tropical_systems

[6] http://www.bom.gov.au/climate/averages/wind/selection_map.shtml

Australia is unique in having warm, poleward-flowing currents along both its east (EAC) and west (Leeuwin Current) coasts, which result in significant tropical communities along both coastlines (Lough 2008). Evidence is emerging for significant changes in the EAC which, over the period 1944–2002, has increased its southward penetration by ~350 km, bringing warmer saltier waters further south (Ridgway 2007, Hill et al. 2008). The intensification of flow and accelerated warming observed in the EAC is driven by the strengthening and contraction south of Southern Hemisphere westerly winds (Poloczanska et al. 2012). Since the mid-1970s, the Leeuwin Current has weakened due to more frequent El Niño events. However, in the past two decades, a strengthening is observed, linked to natural decadal variability and not long-term change (Poloczanska et al. 2012). The intensity of the Leeuwin Current is also significantly modulated by ENSO events, in particular La Niña events are associated with a strengthening of the current and transport of warmer waters further south, as happened in 2011 (Feng et al. 2013) resulting in significant impacts on west coast marine ecosystems (Wernberg et al. 2012).

Ocean circulation patterns are less well documented for the GoC region. In the Gulf of Carpentaria barotropic diurnal tidal currents dominate (Church and Forbes 1983). Observations also show a slow, clockwise circulation, which appears to be a permanent feature in the Gulf. Northwest monsoon winds and density-induced currents enhance the clockwise circulation. However, when the south-east trade-winds build at neap tides they drive a counter-clockwise circulation, and at spring tides, a weak clockwise circulation (Forbes and Church 1983).

8.2.3   Climate projections

The climate projections presented here are based on the IPCC-AR4 CMIP3 global climate model outputs downscaled for Australia (CSIRO and BoM 2007, Lough 2007) and specific regions (Bell et al. 2011a, Poloczanska et al. 2012) for 2030 and 2070 (or the nearest available projection years) for a moderate emissions scenario (SRES A1B/A2) and a high ('business-as-usual') emissions scenario (SRES A1FI) (IPCC 2007). Where available, updated projections for IPCC-AR5 based on the newly developed CMIP5 models are presented (IPCC 2013). This new generation of models operate at higher spatial resolution and include a wider range of climate processes than CMIP3.

The IPCC-AR4 assessments were based on the Special Report on Emissions Scenarios (SRES, Nakicenovik et al. 2000) which have been replaced in IPCC-AR5 with a new set of scenarios: the Representative Concentration Pathways (RCPs; Moss et al 2010). These are named after the level of radiative forcing in 2100, i.e. the change in the balance of incoming and outgoing radiation to the atmosphere due to changes in the atmospheric concentration of greenhouse gases such as CO_2. Although not directly comparable, the SRES A1F1 scenario is very similar to RCP8.5 and the moderate SRES A1B/A2 scenarios are similar to the RCP6 (see

Figure 8.6). Despite the changes in the models, however, climate projections from CMIP3 and CMIP5 do not differ greatly and the basic conclusions from previous assessments remain largely valid (Knutti and Sedláĉek 2013, IPCC 2013).

The projections do not show significant divergence by 2030 under the different SRES or RCP emissions scenarios but do by 2070. The results show that changes in sea surface temperatures, rainfall, sea level, ocean chemistry and salinity are expected to occur, which are likely to impact on biological productivity of marine environments. Although changes in *average* climate conditions are expected to cause major impacts on tropical Australian marine environments, changes to the intensity and frequency of climate *extremes* such as tropical cyclones and floods are likely to be even more significant, as witnessed during the 2012/13 Austral summer (Climate Commission 2013).
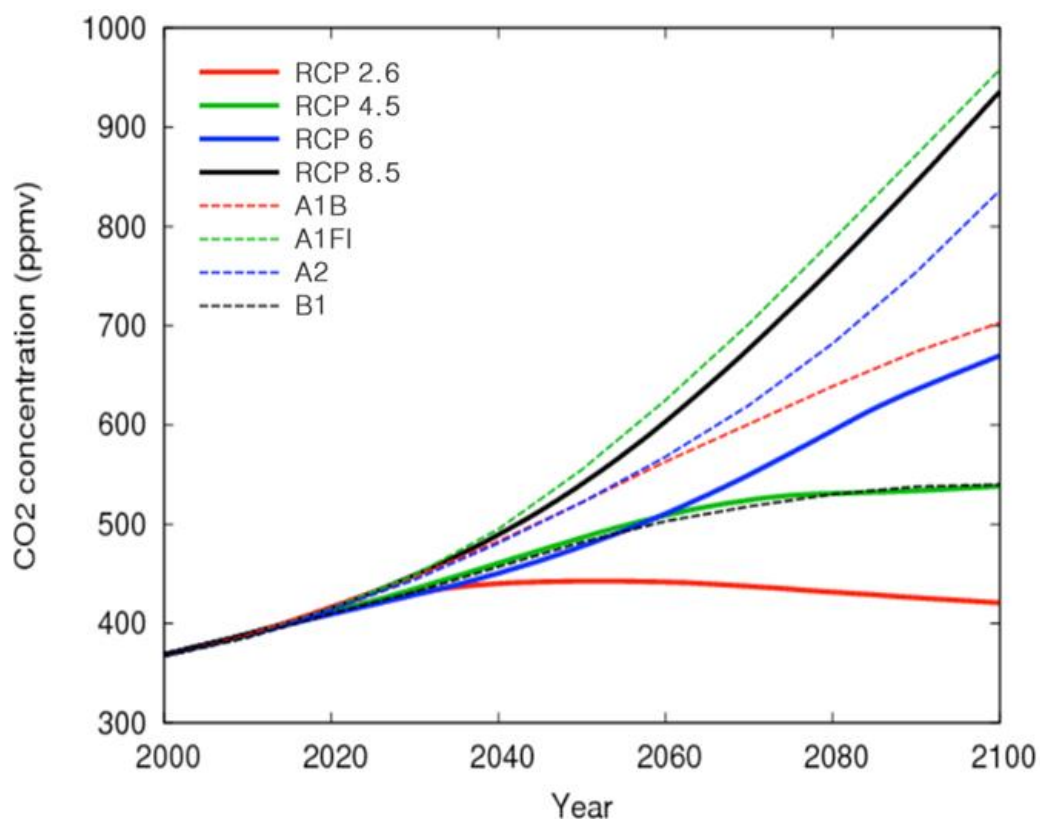
Figure 8.6 Comparison of IPCC AR-4 SRES and AR-5 RCP scenarios for CO$_2$ projections (Collier et al. 2011).

Sea surface temperatures
Sea surface temperatures are expected to continue warming during this century, affecting maximum, minimum and mean temperatures. The projections for sea temperature in northern Australia under the moderate A1B emissions scenario are between 0.3–0.6 °C warming (relative to 1980-1999) by 2030 for the EC and GoC, and 0.6 to 0.9 °C warming by 2030 for NWA (CSIRO and BoM 2007) (Table 8.10). The pattern of greatest SST increase in

northwest WA continues to 2070 under both the moderate and high emissions scenarios (Table 8.10). Results from the newly developed CMIP5 models for the relatively high RCP8.5 emissions pathway (equivalent to A1FI) also indicate greatest tropical ocean warming off northwest Australia of ~2.5 °C by 2100 (Lough et al. 2012).

Table 8.10 Projected increases in sea surface temperatures for northern Australia (CSIRO and BoM 2007).

	2030 (A1B)	2070 (A1B)	2070 (A1FI)
East coast (°C)	+0.3 to +0.6	+1.2 to +1.5	+2.2 to +2.5
Gulf of Carpentaria (°C)	+0.3 to +0.6	+1.2 to +1.5	+2.2 to +2.5
Northwest Australia (°C)	+0.6 to +0.9	+1.5 to +1.8	+2.5 to +2.8

Rainfall and River flow

Projected rainfall changes are more variable and uncertain, with both increases and decreases expected; the EC is projected to become drier while the GoC and NWA are projected to become wetter (Table 8.11). There is also still a high degree of uncertainty associated with rainfall projections for tropical northern Australia where CMIP5 models still have a lack of agreement between models and a wide spread in the simulations (Irving et al. 2012). Extreme rainfall events are projected to occur more frequently with more intense rainfall and more dry days in between (CSIRO and BoM 2007, BoM and CSIRO 2011).

Table 8.11 Projected changes (%) in rainfall for northern Australia (CSIRO and BoM 2007).

	2030 (A1B & A1FI)	2070 (A1B)	2070 (A1FI)
East coast	-10 to 0	-20 to +10	-30 to +10
Gulf of Carpentaria & North-western Australia	0 to + 5	0 to +20	

Predicting river flow changes is largely based on rainfall projections, and the highly seasonal and variable rainfall regime of tropical regions in Australia also results in highly variable river flows (Lough and Hobday 2011). More extreme rainfall events will most likely result in more extreme flood events (Climate Commission 2013).

El Niño –Southern Oscillation

The current generation of global climate models are not good at representing the variability associated with ENSO, and show little consensus on the simulation of likely changes in the frequency, intensity and patterns of future El Niño and La Niña events (Collins et al. 2010, BoM and CSIRO 2011). Therefore, all that can be said about ENSO in the future is that it will continue to be a source of inter-annual variability in the region but super-imposed on warmer SST (Lough and Hobday 2011, Poloczanska et al. 2012).

Tropical cyclones

There is large uncertainty about how tropical cyclones will change under a warmer climate. However, a review of modelled tropical cyclone characteristics predict a likely increase in the maximum intensity of tropical cyclones as the mean global temperature rises, of between +3% to +21% by 2100, or between +2% and +11% if expressed as maximum wind speed (Knutson et al. 2010). The consensus from many advanced modelling studies is that the frequency of tropical cyclones will either stay the same or decrease, ranging from -6% to -34% globally by 2100 (Knutson et al. 2010), and these projected patterns are largely expected to play out in northern Australia (CSIRO and BoM 2007). It is possible that the recent trend of more frequent TCs in NWA will continue. Ultimately, tropical cyclone numbers are projected to decline in the southwest Pacific (BoM and CSIRO 2011) in the future but those that do occur are likely to be more intense (Lough et al. 2011).

Ocean chemistry

Increases in atmospheric CO2 are projected to lead to substantial additional acidification of the ocean, reducing the pH of the ocean by 0.2–0.3 units (under A2) by 2100. At such rates of change, aragonite saturation levels in the tropical Pacific Ocean are expected to fall below 3.5 by 2030 (under A2), jeopardising the growth of corals, shellfish and some plankton. Projections for the mid-term are that ocean pH will decline by 0.1 unit by 2035 under the A2 IPCC-AR4 emission scenario (Ganachaud et al. 2011). The aragonite saturation level is expected to decrease to 2.4 in 2100 (under A2), with severe consequences for the formation of coral reef habitats and many reef organisms (Ganachaud et al. 2011, BoM and CSIRO 2011).

Sea level

The current rate of sea-level rise (1993 to present) is about 3.1 ± 0.4 mm/yr (Church et al. 2012). To date, most of this rise has been attributed to thermal expansion as the oceans have warmed. With continued thermal expansion of the upper ocean layers and a greater contribution from melting of land-based ice, this rate is expected to accelerate. The projections from IPCC–AR4, that sea level will rise between 18 cm (under the B1 low emissions scenario) to 51 cm (under A2) by 2100, are now considered to be conservative because they do not include the effects of increased solid ice flow (Ganachaud et al. 2011). More recent estimates using the CMIP3 models, simulate a sea-level rise of 5 – 15 cm by 2030 and 20 – 60 cm by 2090 (under A2) (BoM and CSIRO 2011). It is also projected that the rate of sea-level rise will be regionally variable, with the GoC and NWA likely to experience a greater increase of 10 – 20 cm by 2030 (Church et al. 2012). Even the lower estimates would mean a profound change for coastal habitats. Confidence in IPCC-AR5 projections of sea-level rise has increased since IPCC-AR4 due to improved understanding and modelling of key processes. IPCC-AR5 projections suggest sea levels at the end of the 21st century (2081-

2100) relative to 1986-2005 are likely to be in the range of 33 to 63 cm for RCP6.0 and 45-82 cm for RCP 8.5 (IPCC 2013).

Rising sea level influences the frequency of extreme high sea-level events (e.g. king tides) that occur on annual to decadal timescales, which has increased by a factor of about three during the 20th century (Church et al. 2012). Higher sea levels would also increase the magnitude and destructive capacity of storm surges associated with tropical cyclones crossing the coast.

Ocean stratification, upwelling and currents
Projected alterations in the speed and direction of some major Pacific Ocean currents – for example, a progressive weakening of the South Equatorial Current (SEC) by 26% by 2100 under A2 (Ganachaud et al. 2011) – will have potential implications for currents, stratification and productivity on the EC and possibly GoC. Changes in the variable and complex tidal regimes of tropical Australia will have implications for species life cycles, and larval and nutrient exchange. The projected increased stratification of the upper layers of the ocean is a major factor influencing the supply of nutrients from the deep ocean to the surface zone and will impact on primary productivity and ultimately fisheries in the region (BoM and CSIRO 2011).

The EAC along the EC is projected to increase in flow off southeast Australia with a compensating decrease off north-east Australia (Ridgeway and Hill 2012). The Leeuwin Current along northwest WA is predicted to weaken over this century. Despite this, warming will continue to drive southward range shifts in marine biota and there will be more frequent extreme temperature events (Ridgeway and Hill 2012).

Ocean salinity and Solar radiation
Other ocean climate variables that are expected to influence fisheries in northern Australia and supporting habitats are ocean salinity and solar radiation. Salinity can have direct effects on fish species and life cycle stages, while solar radiation is important for the growth and maintenance of seagrass meadows, a critical habitat for many fisheries species.

A reduction in salinity, or freshening, has been observed over recent decades in the western tropical Pacific Ocean (Cravatte et al. 2009, Durack et al. 2012). Sea surface salinity is projected to continue to decrease by 0.1 psu (on the practical salinity scale) by 2030, and 0.34 psu by 2090 under the A2 IPCC-AR4 scenario (BoM and CSIRO 2011). Solar radiation is projected to undergo minor changes of -1% to +2% by 2030, with larger changes projected for 2070 however, there is high uncertainty (CSIRO and BoM 2007).

8.2.4   Summary of climate projections

A summary of the climate projections for 2030 and 2070 under the moderate and high SRES emissions scenarios (A2/A1B and A1FI) for key variables is provided in Table 8.12.

Table 8.12 Summary of climate projections for northern (tropical) Australia for 2030 and 2070 under the A2/A1B and A1FI emissions scenarios.

Variable	2030	2070	
	A1B/A1FI	A2/A1B	A1FI
SST (°C)	+0.3 to +0.6 (EC, GoC); +0.6 to +0.9 (NWA)	+1.2 to +1.5 (EC, GoC); +1.5 to +1.8 (NWA)	+2.2 to +2.5 (EC, GoC); +2.5 to +2.8 (NWA)
Ocean temp >250 m (°C)	0 to +0.6[a]	+0.6 to +1.5	+0.6 to +2.4[b]
Rainfall change (%)	-10 to 0 (EC); 0 to +5 (GoC, NWA)	-20 to +10 (EC); 0 to +20 (GoC, NWA)	-30 to +10 (EC); 0 to +20 (GoC, NWA)
Riverflow/nutrient supply	1:4 reduction	Region specific[c]	
ENSO	Continued source of interannual climate variability		
Storms & cyclones[d]	-9 to -44% number[e]; +3 to +21% intensity		
Ocean pH	~7.98	~7.81	
Sea level (cm)	+5 to +15 (EC); +10 to +20 (GoC, NWA)	+20 to +60 (by 2090)	
Ocean circulation	Strengthening of EAC; weakening of Leeuwin current		
Sea surface salinity (psu)	-0.1	-0.34 (by 2100)	

(a) n/a for GoC; (b) northern EC the warmest; (c) linked to rainfall changes; (d) by 2100; (e) possibly more frequent TCs in NWA.

Sources: Climate Change in Australia, OzClim, CSIRO and BoM 2007, Cravatte et al. 2009, Knutson et al. 2010, Bell et al. 2011b, BoM and CSIRO 2011, Lough and Hobday 2011, Church et al. 2012, Lough et al. 2012, Poloczanska et al. 2012.

8.3 Climate change implications for habitats that support northern Australian tropical fisheries

8.3.1 Overview

The natural ecosystems that northern Australian fisheries rely on have evolved to operate within a specific range of prevailing local climatic conditions – a tolerance range (e.g. Jones and Mearns 2005, Hoegh-Guldberg et al. 2007). Changes beyond these specific conditions will influence the habitats that support fisheries, as well as fisheries stocks, species, populations and communities themselves. Tropical fisheries that target species with strong ecological relationships to specific microhabitats or a combination of seasonally-available habitat patches are most likely to be influenced by climate related impacts (Badjeck et al. 2010, MacNeil et al. 2010, Donnelly 2011, Pratchett et al. 2011, Bell et al. 2013).

Understanding how climate change is likely to influence a range of key habitats – coral reefs, seagrass meadows, mangroves, estuaries and floodplains (Figure 8.7) – is critical to assessing fisheries changes under future climate scenarios. The aim of this chapter is to review the range of potential climate change impacts on key fisheries habitats across northern Australia. The project is focused on fisheries across northern Australia covering a vast area over three regions: north-western Australia (northern Western Australia and north-western Northern Territory; NWA), the Gulf of Carpentaria (GoC), and the Queensland east coast (EC). Our review considers the vulnerability of fisheries habitats in these three regions to climate change and what impacts might manifest in the future. Section 8.2 provides climate projections for the three regions of northern Australia for 2030 and 2070 under the IPCC SRES A1B/A2 (moderate emissions reductions) and A1FI ('business-as-usual') scenarios, which are referred to in this review.
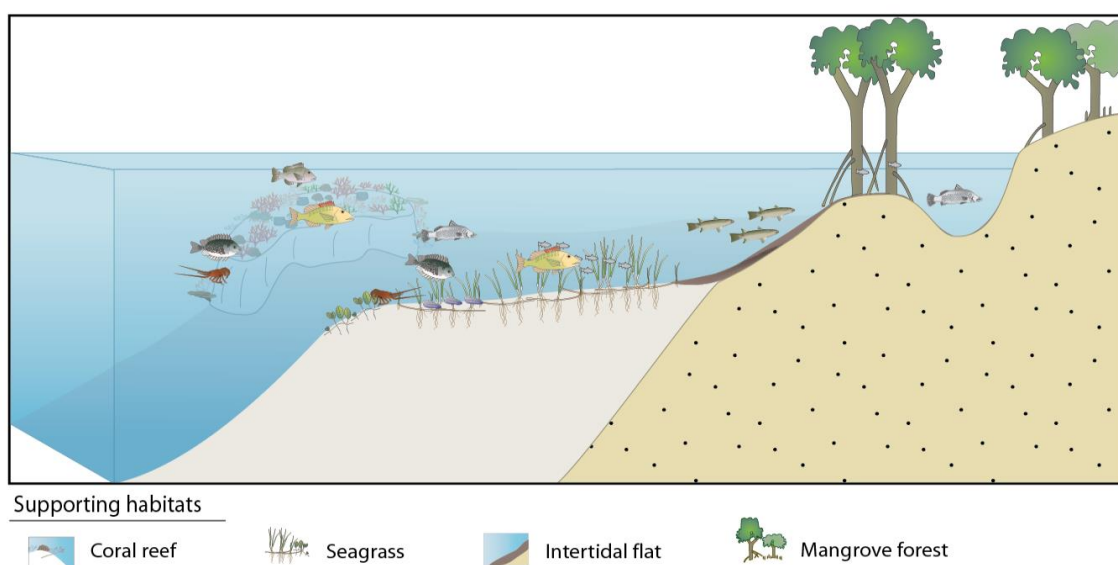
Figure 8.7 Graphic representation of tropical habitats and their connectivity.

8.3.2 Exposure of northern Australian habitats

Marine environments in northern Australia range from floodplains, coastal bays and mangrove-lined estuaries, through near-shore intertidal flats and seagrass habitats, to coral reefs, deep-water seagrass meadows, and wider continental shelf and open-ocean pelagic habitats (Poloczanska et al. 2007). These various habitats are connected by water movements that influence transport of fish larvae, sediment, nutrients and other marine organisms, as well as dynamic temperature and salinity gradients. Tropical fish species utilise these habitats during different life-history stages, and often move between habitats.

Coastal mangrove forests and intertidal flats are found throughout northern Australia, including the EC, GoC and NWA regions of this project, particularly where rivers and estuaries meet the coast (Figure 8.8a). The EC of Australia is characterised by significant coral reef areas with high coral species diversity between latitudes 10°and 25°S (Great Barrier Reef and Torres Strait). While NWA has coastal reefs between latitudes 20° and 24 °S and a concentration of offshore reefs centred around 17 °S (Rowley Shoals; Figure 8.8b).

Coral reefs on the EC are interspersed with shallow seagrass meadows, with an estimated ~35,000 km^2 representing >50% of seagrass area in Australia (McKenzie et al. 2012). In the GoC, the generally shallow and soft sediment environment supports extensive areas of seagrass in coastal and estuarine locations, however recent mapping observed low diversity and biomass[7]. The large tidal variation (1 - 11 m) in NWA causes strong tidal flows that dramatically influence coastal habitats and seagrass meadows are mostly found in sheltered intertidal bays along the southern coast of the Kimberley region, with low to moderate abundance. Seagrasses are also interspersed in coral reef environments in NWA but the high-energy environments of the northern Kimberley means seagrass are largely absent on that part of the coast[8] (Figure 8.8c).

The location of coastal habitats will determine their exposure to projected future climate change: increasing sea surface temperature (SST), ocean acidification, changing rainfall and river flow patterns, sea-level rise, more intense storms and cyclones, and changing ocean circulation. Although all three regions in northern Australia are projected to experience increases in SST, the magnitude of increase will be greatest in NWA meaning that coral reefs and mangrove forests in this region will be exposed to higher sea temperatures. Similarly, habitats in NWA and GoC will be exposed to wetter conditions with rainfall projected to increase, while habitats on the EC will be exposed to drier conditions with rainfall projected to decrease under all scenarios (see Table 8.12 for details of A1B/A2 and A1FI 2030 and 2070 projections).

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[7] http://seagrasswatch.org/Napranum.html
[8] http://seagrasswatch.org/WA.html
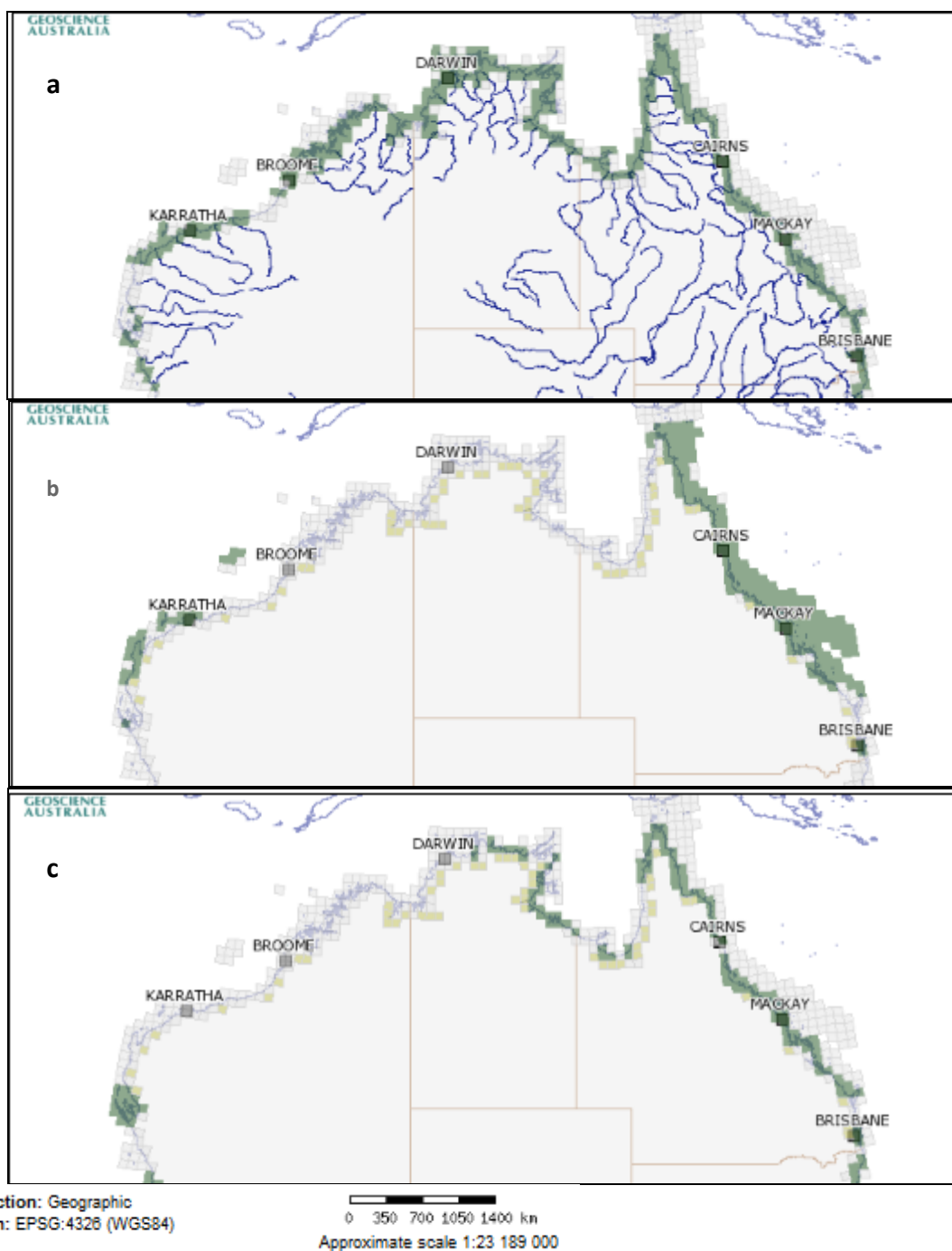
Figure 8.8 Location of marine and coastal habitats in northern Australia: (a) Rivers, estuaries and mangroves, (b) coral reefs, and (c) seagrass meadows (Source: OzCoasts, Geoscience Australia).

8.3.3 Habitat types

Floodplains

Floodplains are shallow, well-vegetated habitats adjacent to lowland river channels. Floodplain habitats are prevalent across northern Australia, occupying over one third of

most river catchments. Most are largely unmodified by human impacts, but they vary in extent and nature. In NWA and GoC, floodplains become available to fish with the onset of the annual flood pulse that inundate this habitat mainly during December-February (Warfe et al. 2011). Extent and intensity of this flood pulse varies significantly across tropical Australia. For example NT Rivers generally flood for a long period of time compared to the Mitchell (GOC) and Fitzroy (EC) Rivers that flood for a shorter period (typically a few days) due to having less extensive catchments (Warfe et al. 2011). During the dry season, as water depth and water quality parameters decline in floodplain waters, the availability and quality of floodplains as fish habitat becomes limited, and fish kills in isolated and drying wetlands are common.

Floodplains provide an array of rich food resources for fish, driven by local algal production. This food supply includes vegetation, insects, crustaceans and juvenile fish, and supports marine fisheries production both directly and indirectly (e.g. as a source of material for downstream habitats). For example, Jardine et al. (2012) examined food web structure in floodplain habitats of the Mitchell River using stable isotopes. They found that floodplain food sources accounted for the majority of the diet of large-bodied fishes captured on the floodplain in the wet season, including barramundi, and for gonadal tissues of a common herbivorous fish (gizzard shad, *Nematalosa come*), the latter suggesting that critical reproductive phases are fuelled by floodplain production. They also found that floodplain food sources subsidised barramundi from the recreational fishery in adjacent coastal and estuarine areas. This increased food, in conjunction with providing shelter from predators, means that floodplains also provide an important nursery habitat for a wide range of fish and invertebrate species (Bunn and Arthrington 2002). This relationship is a key driver for the productivity of important tropical species such as barramundi with recruitment success being driven by floodplain inundation (Robins et al. 2005).

Floodplain ecosystems are sensitive to changes in river-flow regimes that affect the hydrological features of the flood pulse (Bunn and Arthington 2002). Consequently, floodplain habitats are likely to be affected by changes to the climate system that affects timing, duration and magnitude of inundation events, including interactions between rainfall, river discharge and sea level. While there is inherent uncertainty in predicting the ecological effects of such changes on fisheries, previous reviews (principally Pusey and Kennard 2009) have consistently identified two key drivers of change in northern floodplain systems: sea-level rise and changing rainfall patterns.

Sea-levelrise is predicted to increase by 0.6 m by 2090 (BoM and CSIRO 2011). Many northern wetlands are located only minimally above sea level and are at extreme risk from sea-level rise (Low 2011, Pusey and Kennard 2009). Finlayson et al. (2002) predicted that the Alligator Rivers region (NWA) will lose existing mangrove forests, followed by an upstream change in their distribution, with a concomitant loss of Melaleuca wetlands and a

transformation of existing freshwater wetlands to saline flats. These impacts may be amplified by increased severity of monsoonal storms and associated storm surges, with the present 1 in 100 year event potentially occurring more than once a year by 2100 (Church et al. 2008). Modelling of such scenarios indicates that the frequency of saltwater inundation of the Kakadu floodplain (NWA) will increase by 60% in 2030 and 500% in 2070 (BMT WBM 2010).

Changing rainfall patterns (particularly greater variability of rainfall and more extreme events) are expected to have pronounced effects on floodplains through alterations to hydrological regimes (Day et al. 2008). On the EC lower rainfall is likely to result in fewer flood events that will mean shorter inundation periods that may not enable sufficient exchange of biota and materials between habitats. Alternatively the rainfall predicted in the GOC and NW will potentially increase the flood period inundation allowing for greater productivity of biota in catchments in these areas. Increased temperatures are likely to result in increased production and decomposition rates in floodplains (Gehrke et al. 2011). Evaporation rates will also increase significantly as atmospheric temperatures rise and this may impact on both persistence and water quality (e.g. dissolved oxygen concentration) on floodplains. Such changes would impact greatly on species that are obligate floodplain dwellers or use floodplains at critical phases of their life history (e.g. many species of estuarine and freshwater fish; Pusey and Kennard 2009). Overall, the important roles that coastal floodplains play as nursery habitats and for water purification are likely to be compromised, ultimately affecting downstream fisheries.

Coastal bays and estuaries
Coastal bays and estuaries form a transition zone between river and ocean environments and are subject to both marine influences, such as tides, waves, and the influx of salt water; and riverine influences, such as flows of fresh water and sediment. These two influences provide high levels of nutrients in both the water column and sediment, making estuaries among the most dynamic and productive natural habitats in the world. At the interface between land and sea, estuaries will be highly exposed to changing rainfall patterns and river flows, intense storms and cyclones, changes in ocean chemistry, highly variable SST and sea-level rise. However, they are accustomed to large variability in environmental conditions, which may in fact make them less sensitive to changing climate conditions. The potential impacts of climate change, and ultimately the vulnerability of estuaries, will depend on the dominant habitat, since they can be comprised of a range of different habitats, including mangroves, shallow seagrass meadows and intertidal flats.

Estuaries dominated by seagrasses, adjacent to rivers and heavily exposed to increased terrestrial runoff, are likely to have high vulnerability to future changes in rainfall and pollutant runoff, surface temperatures, and physical disturbance from cyclones and storms. While estuaries with mangrove habitats will be vulnerable to more intense storms and

cyclones, changing rainfall patterns and river flow, and sea-level rise, with high sediment accumulation rates allowing some adaptation to rising sea levels (Waycott et al. 2011). More details on the specific impacts and vulnerability of seagrass meadows and mangroves to future climate change are provided below.

Intertidal estuarine habitats will be particularly exposed to rising SST as they experience periods when peak daytime temperatures coincide with low spring tide exposure, resulting in possible losses of intertidal organisms despite the high stress-tolerance of many species (Brierley and Kingsford 2009). This will be particularly pronounced in NWA estuaries, where the greatest SST increases are projected. Increased temperature is expected to potentially inhibit intertidal primary productivity in estuaries (Gehrke et al. 2011).

Estuaries are highly variable habitats and their fauna and flora have evolved to deal with environmental variability. For example, recorded pH in the Fitzroy River estuary (EC; a primary habitat of barramundi) can vary between 8.6 and 6.8 (Robins, unpublished data). The potential impacts of projected pH reductions under climate-change scenarios (0.5 unit decline; Gillanders et al. 2011) are likely to be relatively minor when compared to this natural variation (Meynecke et al. 2013).

Estuaries in low-lying areas are likely to expand inland with rising sea levels, as inundation by freshwater inflows increases during high rainfall periods. Tidal movements and salinity will extend further inland. These effects will be accentuated by storm surges during any cyclones of higher intensity (Gehrke et al. 2011).

Changes to estuarine habitats will have implications for the fisheries they support. For example, examination of NSW commercial fisheries data has shown that catch-per-unit-effort (CPUE) increased in proportion to freshwater flow for four commercial estuary species (dusky flathead, luderick, sand whiting and sea mullet) and decreased during drought (Gillson et al. 2009). Booth et al. (2011) found similar correlations, with increases in overall CPUE of the EC northern mud crab fishery interpreted as a response to SST increases. Barramundi landings have been correlated to an index of climate variability (Balston 2009a), and nursery habitat productivity (Balston 2009b) in estuarine habitats.

Seagrass meadows
Seagrasses provide nursery areas for many commonly harvested fish and invertebrates (e.g. tiger prawns, sandfish and red emperor), and feeding grounds for many species of prey and adult demersal fish targeted by fisheries (e.g. barramundi and black jew). Seagrasses (and intertidal flats) are also permanent habitats for a wide range of invertebrates, such as sea cucumbers.

Seagrasses face an array of pressures as human populations increase and the potential effects of climate change, such as increased storm activity, come into play (Waycott and McKenzie 2010, Grech and Coles 2010, Grech et al. 2011). Changes to nutrient dynamics and light penetration in coastal waters have been documented to impact on seagrass extent and condition with continued declines recorded on the EC since 2005 (McKenzie et al. 2012). Chronic elevated nutrients have been reported to lower the availability of light to seagrasses due to increased growth of algae and epiphytes on the plants (Burkholder et al. 2007). Chronic and pulsed increases in suspended sediments that increase turbidity can also reduce light and result in reduced productivity and potentially seagrass loss (Waycott and McKenzie 2010).

Tropical seagrasses require water temperatures of 25 - 35°C and when SST rises to 35 - 40°C, photosynthesis declines due to the breakdown of photosynthetic enzymes (Ralph 1998) and can result in reduced growth rates (Waycott et al. 2011). Although temperature tolerance varies between species and seasons (Campbell et al. 2006, Perez and Romero 1992), overall seagrass can only survive temperatures >40°C for short periods, and prolonged exposure leads to the 'burning' of leaves or plant mortality (Waycott et al. 2011). Although seagrass meadows in NWA are not an extensive habitat, they will be exposed to a projected SST increase of 2.5 to 2.8 °C by 2070, and may therefore experience earlier or greater impacts.

Severe cyclones and storms physically damage seagrass meadows, particularly in shallow locations (Waycott et al. 2011, McKenzie et al. 2012). For example, seagrass meadows on the EC were impacted by Tropical Cyclone Yasi and associated flooding during the 2010/11 wet season, with 98% of the intertidal seagrass area lost as a consequence of the destructive winds (McKenzie et al. 2012). Although seagrass meadows in northern Australia have been impacted by cyclones for hundreds of years, the projected increase in intensity of these events is particularly concerning, as greater impacts coupled with shortened return intervals are likely to hinder the natural recovery cycle. Therefore, seagrasses are predicted to be moderately to highly vulnerable to future projections of changing rainfall patterns and more severe cyclones and storms.

Overall, tropical seagrasses are expected to be vulnerable to increasing SST (particularly in NWA), reduced light penetration (due to increased turbidity or lower solar radiation), changes to rainfall and increases in cyclone intensity (Table 8.13).

The vulnerability of seagrasses to increasing SST, decreasing light penetration, changing rainfall patterns and possible increases in cyclone intensity is projected to reduce seagrass area, with declines expected under both the B1 and A1FI scenarios in the medium- (2030) and long-term (2070)(Waycott et al. 2011). For the tropical Pacific, declines in seagrass area have been predicted of between 5 and 20 % by 2030 (Waycott et al. 2011), and similar predictions are expected for tropical northern Australia.

Table 8.13 Vulnerability of seagrasses to projected changes in surface and ocean climate (adapted from Bell et al. 2011a).

	Sea surface temperature	Solar radiation	Ocean chemistry	Cyclones & storms	Rainfall patterns	Sea level	Nutrient supply
2030 B1/A1FI	Moderate	Moderate	Very low	Moderate	Moderate	Low	Low
2070 B1	Moderate	Moderate	Very low	Moderate	Moderate	Moderate	Low
2070 A1FI	High	High	Very low	High	High	Moderate	Moderate

Mangroves

Mangroves provide nursery areas for many commonly harvested fish and invertebrates, and feeding grounds for many species of adult demersal fish and invertebrates targeted by fisheries (e.g. emperors, snappers, barramundi, mud crab and prawns). Mangroves have evolved to not only tolerate but to depend on tidal inundation by saltwater. However, they are unable to tolerate complete submersion, and as the frequency and duration of inundation increases, growth of trees will decline and forests may retreat landward unless they are able to migrate onto higher ground (Waycott et al. 2011). Thus areas in northern Australia with low tidal ranges, low rainfall and limited sediment supply are more likely to experience retreat of seaward fringing mangroves as sea-level rises. Compared to areas with high tidal ranges, high rainfall and high sediment supply, which are conditions where mangrove expansion is likely to occur (Lovelock et al. 2007, Steffen et al. 2009, Waycott et al. 2011). This has already been observed in other tropical regions, with the gradual retreat of mangroves in southern Papua New Guinea (PNG) in response to rates of sea-level rise similar to those projected (Valiela et al. 2001), and in Micronesia, where mangrove sediments are not keeping pace with current sea-level rise (Wolanksi et al. 2001).

Landward migration of mangroves is only possible if landward barriers, such as roads, levee banks and developments, don't inhibit movement. Under the B1 and A1FI emissions scenarios in 2030 and 2070, mangroves are projected to be most vulnerable to sea-level rise (depending on the rate of increase), and to a lesser extent increasing cyclone intensity and changes to rainfall (Table 8.14). Ultimately, the vulnerability of mangroves to climate change is projected to reduce mangrove area, with declines becoming greater over time.

Mangroves support significant fisheries resources in northern Australia, with production estimates for fish of 20 - 290 kg per ha, and for prawns 450 - 1,000 kg per ha per year (Lovelock et al. 2007). Along the Queensland coast, as in other locations, mangrove cover is positively correlated with fisheries landings (Blaber 2002, Manson et al. 2005). Therefore,

any decline in mangrove area, or loss of connectivity with other critical fish and invertebrate habitats, such as floodplains likely to result in reduced fisheries catches.

Table 8.14 Vulnerability of mangroves to projected changes in surface and ocean climate (adapted from Bell et al. 2011a).

	Sea surface temperature	Surface salinity	Ocean chemistry	Cyclones & storms	Rainfall patterns	Sea level	Nutrient supply
2030 B1/A1FI	Very low	Low	Very low	Moderate	Low	High	Low
2070 B1	Very low	Low	Very low	Moderate	Moderate	Very high	Low
2070 A1FI	Very low	Low	Very low	Moderate	Moderate	Very high	Low

Coral reefs

Coral reefs are an important coastal and offshore habitat in the NWA and EC regions of northern Australia, with thousands of fish and invertebrate species associated with the structures created by corals, several of which have been identified as priority species for this project. Coral reefs support important fisheries for demersal fish (e.g. coral trout, red throat emperor), some near shore pelagic fish (e.g. species of mackerel, sharks), and invertebrates targeted for export and recreation (e.g. tropical lobster, black teatfish). Maintaining the structural complexity of reef frameworks is vitally important to the continuation of these fisheries.

Ultimately, coral reefs are most vulnerable to increasing SST and ocean acidification. Coral reefs are highly vulnerable to further increases in SST due to coral sensitivity to thermal stress, with coral bleaching impacts already documented for most reefs in Australia and around the world as a result of extended periods of above average SST (Wilkinson et al. 2008). The projected increase in SST in northern Australia will influence the structure and function of coral reefs, particularly in NWA where SST increases of 2.5 to 2.8 °C by 2070 are projected (Lough et al. 2012) and isolated offshore reefs can take decades to recover (Smith 2008). Effects will be evident by 2030, with annual bleaching conditions associated with atmospheric CO_2 equivalent concentrations of 510 ppm (under RCP6.0 equivalent to A1FI). Bleaching also shows a latitudinal gradient with higher latitude reefs projected to experience bleaching conditions later under RCP6.0 (equivalent to SRES A1FI)(van Hooidonk et al. 2013).

Ocean acidification is expected to increasingly slow the rate of reef accretion and enhance erosion over the coming decades (Silverman et al. 2009). Reductions in calcification rates at lower ocean pH suggests that corals, and the reefs they build, are highly vulnerable to ocean acidification, and that increases in atmospheric CO_2 above 450 ppm are likely to result in net erosion of coral reefs throughout the tropics (Bell et al. 2011a). A decline in coral

calcification on the GBR was documented by De'ath et al. (2009) and postulated to be due to increasing temperature stress and a declining saturation state of seawater aragonite, with a tipping point reached in the late 20th century. Further, studies in natural CO_2 seeps in PNG (Fabricius et al. 2011) have observed reductions in coral diversity, recruitment and abundance of framework building corals, and shifts in competitive interactions between taxa as pH declines from 8.1 to 7.8 (the change expected by 2100 if atmospheric CO_2 concentrations increase from 390 to 750 ppm). However, coral cover remained constant between pH 8.1 and ~7.8, as massive *Porites* corals dominated, despite low rates of calcification, and reef development ceased below pH 7.7.

Under the B1 and A1FI emissions scenarios in 2030 and 2070, coral reefs are projected to be vulnerable to increasing SST, ocean acidification, and cyclone intensity, as well as ocean circulation and upwelling (Hoegh-Guldberg et al. 2011). The vulnerability of coral reefs to the projected changes in climate is summarised in Table 8.15.

Table 8.15 Vulnerability of coral reefs to projected changes in surface and ocean climate (adapted from Bell et al. 2011a).

	Sea surface temperature	Ocean chemistry	Cyclones and storms	Rainfall patterns	Sea level*	Ocean circulation
2030 B1/A1FI	High	High	Moderate	Moderate	Low	Moderate
2070 B1	Very high	Very high	High	High	Low – Moderate	Moderate
2070 A1FI	Very high	Very high	High	High	Low – Moderate	Moderate

* Range of vulnerability reflects the significant uncertainty regarding the rate of sea-level rise.

The range of potential impacts resulting from future climate change means that coral reef habitats are projected to change, with coral cover expected to decline under both scenarios in the medium- (2030) and long-term (2070), and macroalgae (fleshy and turf algae) projected to become more dominant (Hoegh-Guldberg et al. 2011). Recent modelling showed that at CO_2 levels above ~600 ppm there is a regime shift to alternate coral-algal states, leading to macroalgal dominance at the highest CO_2 level (Anthony et al. 2011). And a long-term study in the Indian Ocean detected declines in reef fishery catches consistent with lagged impacts of habitat disturbance (Pistorius and Taylor 2009). These examples demonstrate the dynamic nature of coral reefs, and how declining reef cover and diversity is likely to have significant implications for fisheries.

Coral reef fisheries are also likely to be affected by predicted reductions in population connectivity due to the effects of climate change on reproduction, larval dispersal and

habitat fragmentation, potentially affecting catch rates and species availability as reef fish community composition changes (Munday et al. 2009).

8.3.4   Conclusions

In northern tropical Australia there is growing evidence of ecosystem and species vulnerability to climate change that has implications for fisheries. Responses to increasing sea surface temperatures (e.g. coral bleaching and mortality, Veron et al.2009), ocean acidification (e.g. reduced coral calcification, De'ath et al.2009; altered reef community structure, Fabricius et al. 2011) and indirect climate effects provide examples of how tropical habitats might change in the future.

Tropical marine and coastal habitats that are subject to local pressures are likely to be more vulnerable to increasing climate change impacts in the future (Veron et al. 2009, Waycott et al. 2009, Anthony et al. 2011, Bell et al. 2011b). Conservation of these habitats (e.g. coral reefs, mangroves and seagrass) has therefore been identified as important to protect important fish species, create natural barriers against sea-level rise and storms, and effective catchment management to minimise impacts from terrestrial runoff on coastal habitats that support coastal fisheries species (e.g. barramundi, prawns)(Holbrook and Johnson 2012, Bell et al. 2011b, Bell et al. 2013).

Table 8.16 Summary table of potential impacts of climate change on northern Australian fisheries habitats by 2030 under the A1B/A1FI emissions scenarios.

Habitat	Region/s	Key potential impacts of climate change	Source
Floodplains	NWA, GoC, EC	Increased temperatures may increase productivity and decomposition rates (+); Changes to rainfall patterns likely to result in more variability in river-floodplain connectivity (+/-); Sea-level rise and storm surge likely to increase salinity inundation and loss of freshwater floodplain habitat area (-)	Gehrke et al. 2011; BMT WBM 2010
Coastal bays and estuaries	NWA, GoC, EC	Increased SST may inhibit intertidal primary productivity (-); Changing rainfall patterns and storm inundation may result in inland area expansions (+); More intense storms and cyclones may alter habitat dynamics and connectivity (+/-)	Gehrke et al. 2011
Seagrass meadows	GoC, EC (small extent NWA)	Increased cyclone intensity and extreme riverflow events may cause extensive localised damage to seagrass beds (-); Reduced solar radiation combined with turbidity from river runoff and storm events is likely to reduce seagrass area available as shelter and food (-) and species diversity (-)	McKenzie et al. 2012; Waycott et al. 2011
Coral reefs	NWA, EC	Increasing SST and SST extremes will likely cause more coral bleaching events resulting in more algal-dominated reef areas (-); Ocean acidification will reduce coral growth and structural integrity and when combined with more intense storms, significant coral loss (-); Combined impacts will result in loss of reef diversity & structure (-)	Veron et al. 2009; Hoegh-Guldberg et al. 2011; van Hooidonk et al. 2012
Mangroves	NWA, GoC, EC	Sea-level rise will result in retreat of seaward fringing mangroves and possible area reductions particularly where there are barriers for mangrove landward migration (e.g. coastal development, sea walls) (+/-); Loss of coastal mangroves combined with more intense storms will result in reduced coastal protection (-)	Ellison et al. 2011; Lovelock et al. 2007

8.4   Sensitivity data analyses

8.4.1   Species and likely environmental driver scoping

For the species-specific data analyses the initial step was to determine the likely drivers of influence for key species. The specific results for each species examined are provided in Appendix 6, while the summary for all species examined is given in Table 8.17 below. The species examined were based on the prioritised lists developed for each region however were also limited to those species that researchers thought potentially had sufficient data for analyses. This process used the published knowledge collated during the individual species reviews, however was largely 'expert' based meaning that most of the results are inferred based on experts knowledge of the particular species and/or knowledge of other species with comparable life histories and habitat preferences. In fact, this process highlighted the complete lack of published knowledge on the sensitivity to climate variability and environmental variables of the vast majority of key fishery species in northern Australia (see Appendix 6).

Due the nature of the framework used, the species judged to be affected the most, and the environmental variables deemed to have the most influence, was largely a reflection of the focus of past research. Although this process is not very conservative (i.e. the sensitivity scores, e.g. SST vs. recruitment, tend to be lower when effects are unknown), it nevertheless provided a basis for further analysis where the certainty in a potential impact on a species is highest. The species chosen for analyses were based on this process including, along with the species reviews, the hypotheses to be tested for each respective species. Across the 19 species examined in this process, changes in SST were considered most likely to have an impact, while nutrients were also important.

Table 8.17 Summary table of the inferred effects of changes in key environmental variables on selected northern Australian fishery species. This was an initial screening process for determining the species for further data analyses and the possible hypotheses for testing. The likely effects of each variable on each species are described as high (H), Medium (M) and Low (L) based on scoring described in Section 6.8.

Common name	SST	rainfall	riverflow	salinity (surf.)	nutrients	upwelling	wind/ currents	pH	sea level
Grey mackerel	H	M	M	M	H	M	L	L	L
Tropical lobster	H	L	L	M	M	M	M	L	L
Coral trout	H	L	L	L	H	M	M	L	L
Spanish mackerel	H	M	M	L	M	M	M	L	L
Red throat emperor	H	L	L	L	H	M	L	L	L
Barramundi	H	H	H	H	H	H	L	L	M
Banana prawn	H	H	H	H	H	H	H	L	M
Scallops	M	L	L	L	L	L	M	L	L
Mud crab	H	H	H	H	M	H	M	L	M
Eastern king prawn	M	L	M	H	L	L	M	L	L
Tiger prawn	H	M	H	H	L	L	L	L	L
Goldband snapper	L	L	L	L	M	M	L	L	L
Red spot king prawn	M	L	L	L	M	M	L	L	L
Sandfish	M	L	L	H	H	L	L	M	L
King threadfin	M	M	M	L	M	L	L	L	L
Golden snapper	L	M	M	L	M	L	L	L	L
Black jew	L	M	M	L	M	L	L	L	L
Scalloped hammerhead	M	L	L	L	L	L	L	L	L
Blacktip sharks	M	L	L	L	L	L	L	L	L

8.4.2 Barramundi

8.4.2.1 Catch data analysis

Both commercial and FTO CPUE was significantly correlated to the number of days that river height was greater than 10m and water year rainfall (Table 8.18, Figure 8.9).

Table 8.18 Pearson correlation coefficient (r) and significance (p) values for annual CPUE for commercial and FTO sectors plotted against annual river height and rainfall environmental variables.

Sector	Environmental variable	r	p
Commercial	River height	0.55	<0.02
Commercial	Rainfall	0.46	<0.05
FTO	River height	0.67	<0.01
FTO	Rainfall	0.55	<0.02

The GLM for the commercial catch showed that fishing effort explained 21.7% of the variation in the catch. The catch adjusted for effort was significantly correlated with the river height and river height 1 variables (Table 8.19). This model explained an additional 27.5% to the base model (total of 49.2% explained). A similar analysis of the FTO catch showed that effort explained a significant proportion (75.6%) of the variation in the catch (r= 0.77, p<0.01). The catch adjusted for effort was significantly correlated with river height (Table 8.19). This model only explained an additional 11.7% to the base model (total of 87.3% explained) indicating that while this was a better model than for the commercial catch the environmental variables were less important in explaining variations in catch.

Table 8.19 Best all sub-sets regression for annual barramundi catch for commercial and FTO sectors and annual river height and rainfall environmental variables.

Sector	Regression model	Percentage variation accounted for (adjusted R^2)
Commercial	River height, River height 1	49.2
FTO	River height	87.3
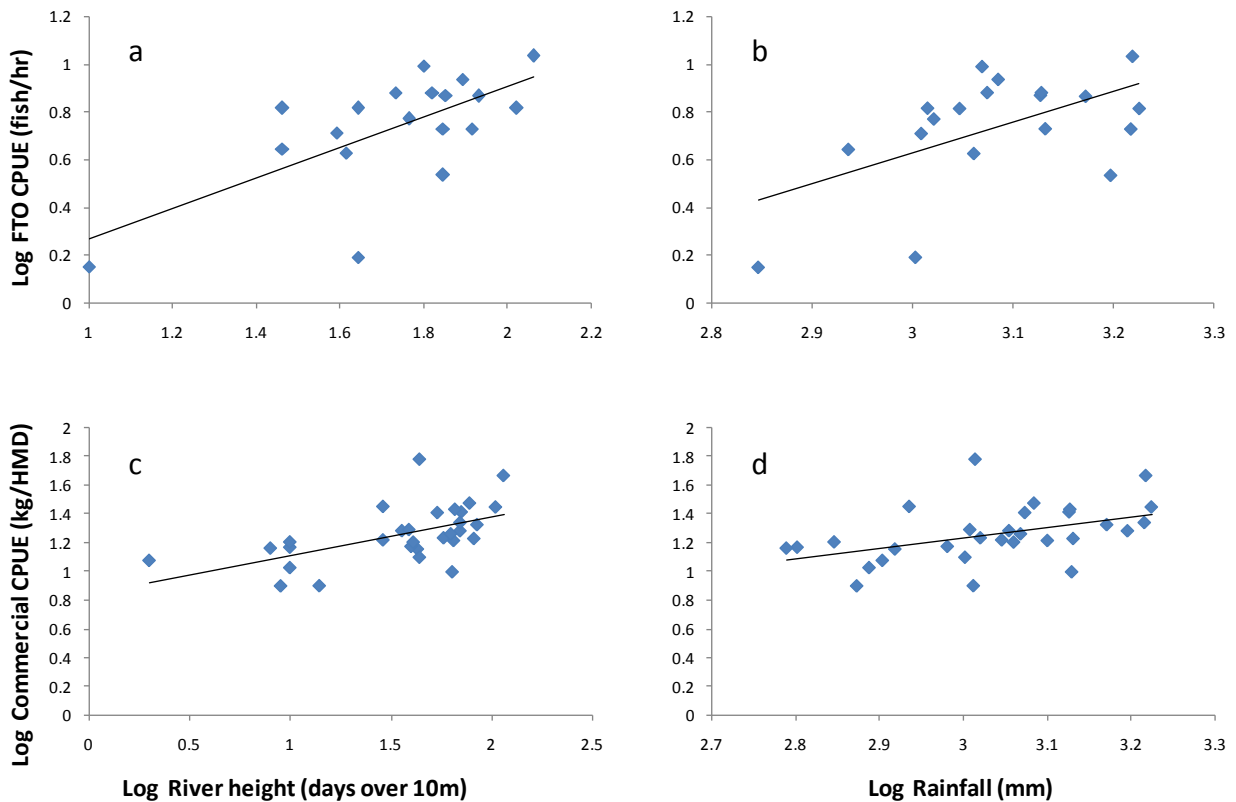
Figure 8.9 Plot of log CPUE against log river height for FTOs (a) and the commercial sector (b) and log rainfall for FTOs (c) and the commercial sector (d).

8.4.2.2 Year Class Strength analysis

Barramundi in the Daly River showed two separate cohorts with a systematic change in their age structure from three to six year olds between 2007 and 2010, and two to five year olds between 2008 and 2011 (Figure 8.10).

The average YCS showed positive residuals for year-classes 'born' in 2004 and 2006 indicating 'stronger' recruitment (Figure 8.11). However there were no large, negative residuals (i.e., <-0.5) indicative of 'weak' barramundi recruitment in the Daly River for the time series examined, and in fact all other years had relatively even residuals (i.e., between +0.8 and -0.4) and thus could not be classified as either 'strong' or 'weak' (Figure 8.11). While the data suggests that river height better explained variability in year class strength (Figure 8.11a) compared to rainfall (Figure 8.11b), neither environmental variable consistently matched the YCS for all years. Consequently, the GLM indicated that neither of these variables explained a significant proportion of variation in YCS (r=0.19, p>0.05).
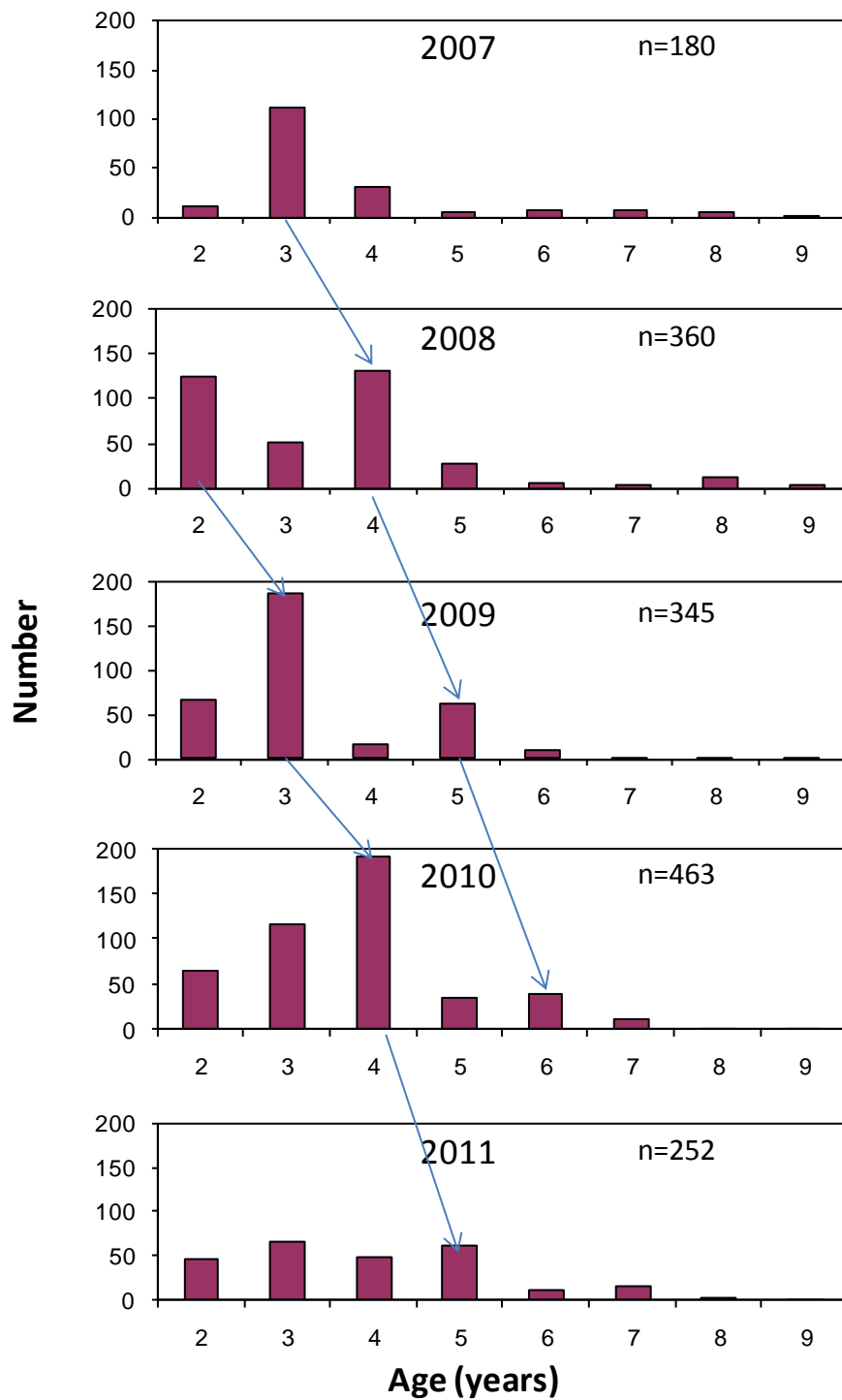
Figure 8.10 Annual age structures of barramundi in the Daly River.
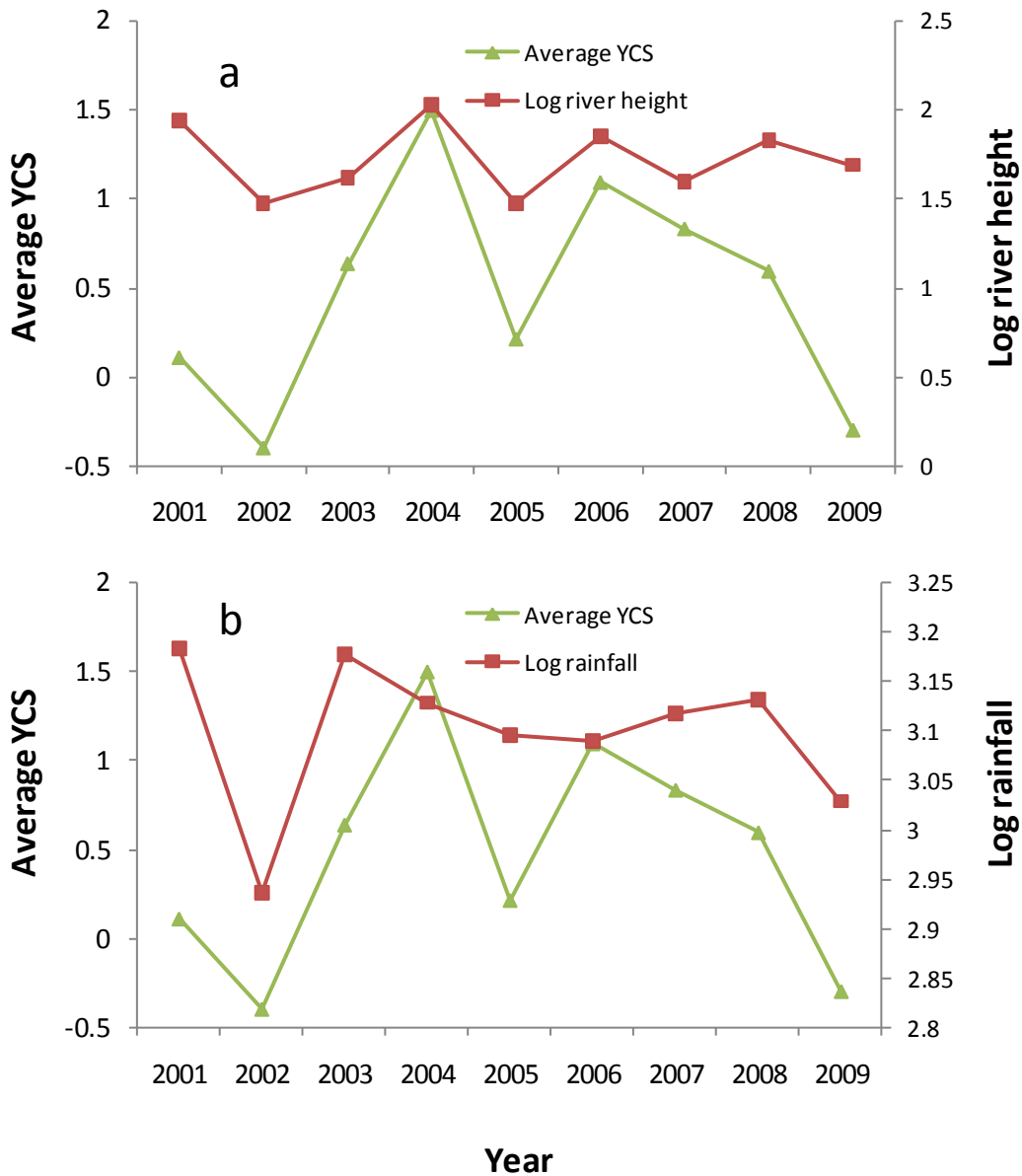
Figure 8.11 Average YCS plotted against (a) log river height and (b) log rainfall during 2001-2009.

8.4.2.3 Discussion

Variations in barramundi CPUE were significantly correlated to variations in both river height and rainfall on the Daly River and this was consistent for both the commercial and FTO sectors. However river height data had a better correlation and the GLM suggested that only river height significantly influenced variations in catch. The reason for this better explanation by river height is that it is more likely to be correlated to flood plain inundation compared to rainfall. The reasons for flood plain inundation increasing barramundi catch have been well documented in other studies (e.g. Robins et al. 2005; Meynecke et al. 2011,

Halliday et al. 2012). Briefly, flood plain inundation promotes production of large volumes of prey items which barramundi aggregate to feed on at the mouth of the river and associated tributaries. These aggregations are easier to target by fishers, which in turn, results in higher catches. The one-year lag in river height that was included in the commercial catch data model suggests that the previous year's productivity on the flood plain has increased growth of barramundi allowing for the total tonnage to increase in the following year. If this was related to recruitment the lags would be at least two years prior to the current year as barramundi only become susceptible to commercial gillnets at between 2-3 years of age (NT Government 2012). It was also interesting that the data suggested that FTO catch had a much higher proportion of its variation explained by effort compared to the commercial catch. This result can probably be explained by the fact that FTOs tend to target 'good fishing' periods during the good wet season years to take clients out, whereas commercial fishers are more likely to fish most of the time during any given year as they rely on catch as income.

Variation in river height and rainfall variables showed some correlation between average YCS, however these relationships were not significant. Significant relationships between barramundi YCS and rainfall and river flow have been found in numerous catchments across northern Australia (Staunton-Smith et al. 2004; Halliday et al. 2011) including the Daly River (Halliday et al. 2012). The lack of a significant correlation between these variables in the current results is probably related to the fact there has been very little contrast in the YCS over the study period with most years having neither strong nor weak recruitment. Where there were significant positive recruitment years there were concurrent increases in the environmental variables suggesting that more years of data would reveal a significant relationship when there is better contrast in the model. Again, the reasons for rainfall and flow variables driving YCS are well explained in the references above. Briefly, increases in these variables are thought to increase recruitment by enhancing the access of larvae and post-larvae to suitable nursery habitats providing increased food availability through higher production of prey items.

Although the results in this particular study were not as equivocal as other recent similar studies regarding environmental influences on recruitment, the results found that higher river height correlated to higher fishery catches. This further adds to the compelling evidence across numerous studies, that barramundi populations and fisheries of northern Australian are significantly influenced by local hydrological characteristics of coastal systems.

8.4.3   Coral trout

8.4.3.1 Year class strength analysis

Age at recruitment to the sampling gear (standard commercial fishing gear) was clearly consistent among regions at age 4 (Figure 8.12) and population age structures are provided in Table 8.20. Close examination of the age frequency distributions by region suggests that *P. leopardus* in the Mackay region may experience higher rates of natural mortality than those populations within either Townsville to the north or Storm Cay to the south. The modal peak at age 4 is strongest for Mackay and the relative strength of progressively older age classes in Mackay decreases faster than either Townsville or Mackay. In addition, the age structure data also contains some evidence of strong year class dominating progressive age classes through consecutive sampling years. For example, in Townsville fish aged 3 dominated the sample in 1997, fish aged 4 dominated the sample in 1998, and fish aged 5 dominated the sample in 1999 (Table 8.20). Notably, no similar trend was observed in either Mackay or Storm Cay. Although weak evidence, these differences in age structure among regions may be reflective of YCS patterns that vary regionally. Thus regional treatment of age structure data was maintained for analysis of year class strength.
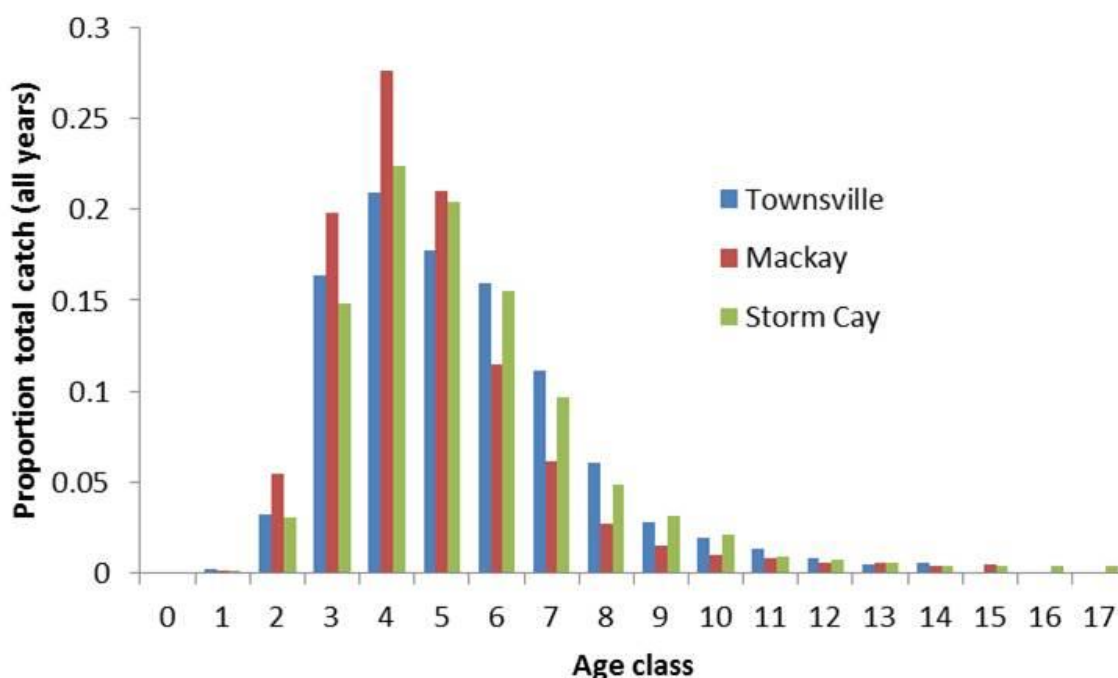
Figure 8.12 Regional age structure of *P. leopardus* demonstrates a consistent age of recruitment to the fishery among the three sampled regions. Age data is pooled across all years.

It was possible to reconstruct the relative year class strength of *P. leopardus* over a 17-year period 1985-2001 (Figure 8.13). Inter-year variability was significant for all regions (ANOVA; F= 3.15, df= 17206, p< 0.001) implying that recruitment was not constant over time. As YCS

was found to be significantly variable through time for each region, analysis of the correlations that may be present between YCS and environmental factors was justified.

The linear mixed-effect models regressing single environmental variables against YCS were mostly statistically non-significant ($p<0.01$) (Table 8.21). In the most southern region of Storm Cay, no significant effect was present for any of the variables tested. Although annual and wet season flows were significantly correlated with YCS in both Townsville and Mackay regions, the effects were not consistent among the in-built lag factors. For example wet season flow from the Burdekin River was significantly correlated with YCS in the Townsville region lagged by one year. In the Mackay region there was a significant correlation between wet season flow from the Fitzroy River and YCS advanced one year. Similar inconsistencies were observed for SST and SOI with both environmental factors detected to be significant in only Townsville (with a 1 year lag) and Mackay (no lag) respectively.
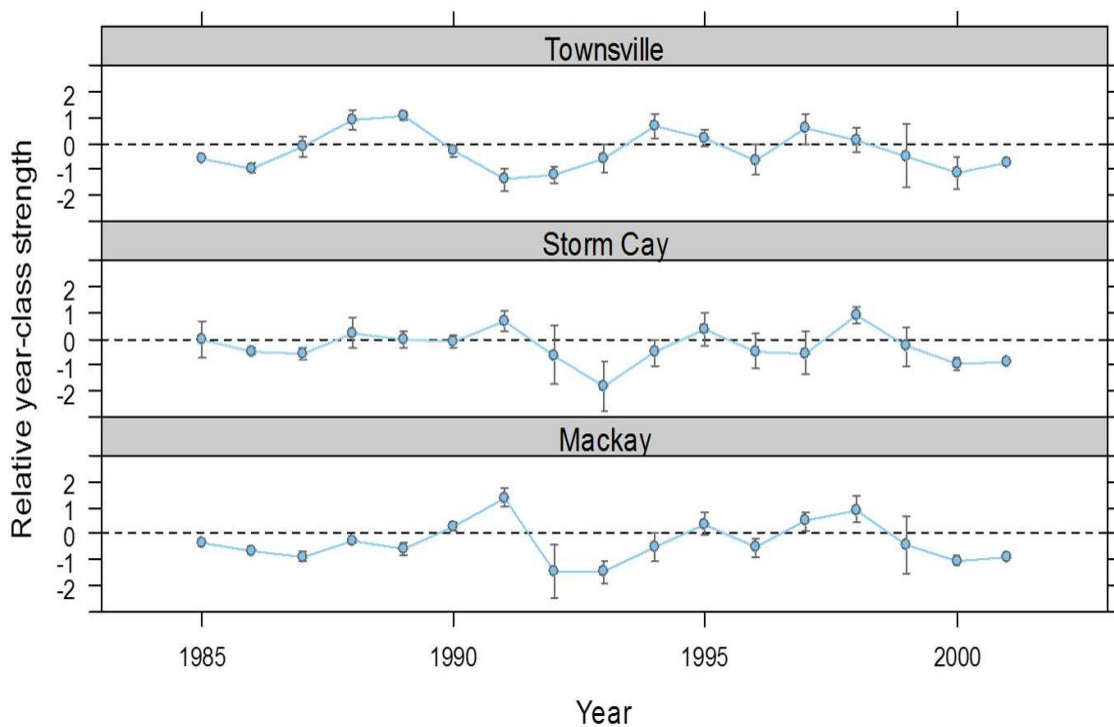
Figure 8.13 Relative year class strength of *P. leopardus* within each of the three sampled regions of the GBRMP.

Table 8.20 Population age structures collected for each year for each region by the Effects of Line Fishing Project. Modal age classes for each year are in bold.

Region	Age	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
TOWNSVILLE	0	0	0	0	0	0	0	0	0	0	0	0
	1	1	0	0	1	1	0	0	1	0	0	1
	2	8	6	23	8	6	7	4	10	1	3	2
	3	25	33	**106**	83	46	26	21	21	5	15	20
	4	47	19	53	**137**	79	56	48	**43**	4	11	15
	5	39	34	27	50	**94**	**74**	42	33	7	20	13
	6	85	28	9	12	41	62	**60**	37	**9**	**23**	**21**
	7	**117**	**37**	10	4	9	12	23	23	6	15	12
	8	48	20	15	8	4	4	2	11	4	7	19
	9	13	5	10	7	1	1	1	4	3	13	3
	10	14	3	4	3	7	2	0	1	0	1	4
MACKAY	0	0	0	0	0	0	0	0	1	0	0	0
	1	1	1	1	0	1	0	0	0	1	0	1
	2	12	30	27	11	14	23	22	11	17	9	21
	3	42	56	**59**	**105**	70	62	116	**61**	28	28	**88**
	4	**229**	104	20	104	**101**	**98**	**137**	47	44	**42**	72
	5	116	**145**	34	38	54	93	110	43	**46**	36	43
	6	46	42	25	13	22	33	95	19	31	35	50
	7	30	20	10	22	18	8	23	17	16	17	35
	8	12	7	2	10	18	6	6	4	7	4	16
	9	7	2	3	4	6	6	5	1	2	4	6
	10	4	0	0	0	3	2	6	1	3	4	3
STORM CAY	0	0	0	0	0	0	0	0	0	0	0	0
	1	0	1	0	0	1	0	1	1	2	0	1
	2	4	27	12	11	13	7	19	13	5	4	9
	3	28	55	48	**91**	121	49	52	**57**	**34**	22	42
	4	**133**	100	36	73	**163**	**124**	89	56	33	30	**65**
	5	117	**138**	**80**	39	74	115	**128**	24	**34**	28	45
	6	83	77	55	63	26	67	105	18	30	**43**	55
	7	69	57	25	39	29	30	47	5	24	23	36
	8	22	22	11	12	26	14	21	6	16	16	25
	9	17	14	6	9	12	12	15	4	6	18	7
	10	16	8	5	6	5	5	10	1	1	10	11

Table 8.21 Results of mixed effect linear regression of environmental variables against recruitment for *P. leopardus* for each of the three sampled regions within the GBRMP. Offset refers to the lag (-1) or advance (+1) in regressing the time series of environmental variables against the time series of year class strength data. The +/- indicates whether the regression relationship is either a positive or negative one.

TOWNSVILLE	Offset	+ / -	F	df	p
SST annual	0	-	0.00	1,60	0.966
	-1	-	0.01	1,57	0.918
	+1	+	4.28	1,61	0.043
SST Sep-Nov (spawning season)	0	+	0.24	1,60	0.626
	-1	+	0.55	1,57	0.547
	+1	+	1.85	1,61	0.179
SOI annual	0	+	0.84	1,60	0.363
	-1	-	0.03	1,57	0.862
	+1	*+*	*10.79*	*1,61*	*0.002*
Burdekin flow annual	0	-	3.13	1,60	0.082
	-1	*-*	*8.10*	*1,57*	*0.006*
	+1	+	0.44	1,61	0.512
Burdekin flow (wet season)	0	+	0.05	1,60	0.833
	-1	-	3.95	1,57	0.052
	+1	*+*	*8.84*	*1,61*	*0.004*

MACKAY	Offset	+ / -	F	df	p
SST annual	0	+	1.31	1,62	0.257
	-1	-	0.23	1,61	0.636
	+1	+	5.03	1,63	0.028
SST Oct-Dec (spawning season)	*0*	*+*	*8.36*	*1,62*	*0.005*
	-1	-	0.11	1,61	0.737
	+1	-	0.01	1,63	0.905
SOI annual	0	-	0.00	1,62	0.950
	-1	+	0.57	1,61	0.454
	+1	+	1.13	1,63	0.293
Fitzroy flow annual	*0*	*+*	*17.13*	*1,62*	*0.001*
	-1	-	1.50	1,61	0.226
	+1	+	0.66	1,63	0.418
Fitzroy flow (wet season)	0	+	0.53	1,62	0.469
	-1	*+*	*13.28*	*1,61*	*0.001*
	+1	-	0.63	1,63	0.430

STORM CAY	Offset	+ / -	F	df	p
SST annual	0	+	2.08	1,62	0.041
	-1	-	0.58	1,61	0.447
	+1	+	0.55	1,63	0.461
SST Oct-Dec (spawning season)	0	+	5.24	1,62	0.025
	-1	+	0.33	1,61	0.565
	+1	-	0.56	1,63	0.457
Fitzroy flow (wet season)	0	+	1.10	1,62	0.298
	-1	+	0.00	1,61	0.992
	+1	+	2.39	1,63	0.127
SOI annual	0	+	4.93	1,62	0.030
	-1	+	0.05	1,61	0.819
	+1	+	0.12	1,63	0.731
Fitzroy flow annual	0	+	0.14	1,62	0.771
	-1	+	3.17	1,61	0.080
	+1	+	0.16	1,63	0.694

8.4.3.2 Discussion

The significant correlations found between YCS and environmental variables were inconsistent among areas, variables and lags in variables. Consequently, no individual variable could be confidently attributed to driving variations in YCS. A number of explanations are possible though insufficient data may be the most plausible. This statement is based upon the acknowledged presence of recruitment spikes (strong year classes) in regional populations of *P. leopardus* within the GBRMP (Ayling et al., 1992; Russ et al., 1996; Doherty et al 1996; Welch 1996). It is possible that data sets that encompass a much longer time series of both age structure and environmental variables than were available for this study are needed to explore the full influence of environmental variables on YCS and hence fisheries productivity.

P. leopardus also live in a relatively stable environment compared to more nearshore habitats (in terms of large annual variation in environmental variables). Compared with shallow coastal and estuarine waters, the water that surrounds and influences fish that live on emergent coral reefs is contrastingly stable. Further, the variation in recruitment that has been observed and described for *P. leopardus* in historical studies (Ayling et al., 1992; Russ et al., 1996; Doherty et al 1996; Welch 1996) may be driven by fine scale changes that are difficult to pin-point in relatively short data time series, or in data that are averaged across too coarse time and/or space units.

While *P. leopardus* inhabit a relatively stable environment they are exposed to cyclones, which have been demonstrated to have a significant influence on their catchability (Tobin et al. 2010). This research described significant reductions in fishery CPUE and landed catch across broad areas of the GBRMP following the passage of severe Tropical Cyclones Justin (March 1997) and Hamish (March 2009). In both events, cool water anomalies were experienced across the fishing grounds and may have been affecting catchability as well as productivity through negative disruptions on growth or mortality of juveniles spawned in the years immediately prior to TC impact. However, in this study the time series of data was too short to attempt an evaluation of infrequent events such as Tropical Cyclones on YCS. However, the impact of cyclones may be masking the effect of the other environmental variables examined due to their significant influence on *P. leopardus* catchability and productivity.

8.4.4 Golden snapper

8.4.4.1 Tagging data analyses

From 1985-2013 there were a total of 31,581 tag days and 105,826 fish tagged in estuary and nearshore habitats of the study area (zones 1-8). While the total number of fish tagged has increased since 2004, the effort expended to do this has been fairly consistent through the time series (Figure 8.14).
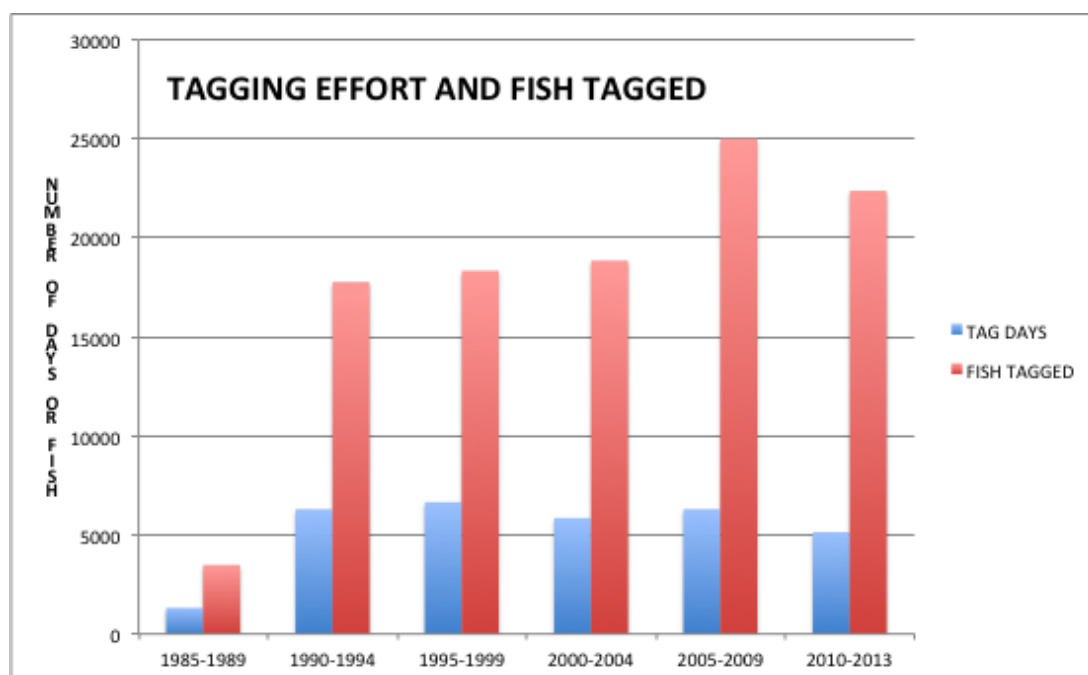
Figure 8.14 Total tag days and fish tagged each 5 years from 1985-2013 from 22°-26° S.

The number of golden snapper tagged as a percentage of the total fish tagged and tagging effort has steadily increased over time (Figure 8.15 & 8.16 respectively). No golden snapper were tagged in zones 3 or 4 until 1990 and none in zones 5 or 6 until 2000. No golden snapper were tagged in zones 7 or 8 over the entire period from 1985-2013. There was an increase in the percentage of golden snapper tagged in zones 1, 2 and 4 (Figure 8.15).

Golden snapper tend to be found in the lower parts of estuaries making the total fish tagged and tagging effort an overestimate of total tagging. However as some golden snapper have been tagged in most parts of the estuaries it was considered that all estuary tagging needed to be taken into account. A number of factors could have contributed to the changes observed. Targeting of golden snapper, and other species, with soft plastic lures has increased, particularly since 2010. However, the trend in increased percentage of golden snapper tagged, was apparent prior to that time. The result could also be a reflection of increased targeting of golden snapper. As golden snapper became more prevalent in the

catch this likely resulted in increased targeting. Increased catch was also likely to have increased skills in catching golden snapper and GPS and new sounder technology have made finding the fish easier, particularly in turbid nearshore habitats.
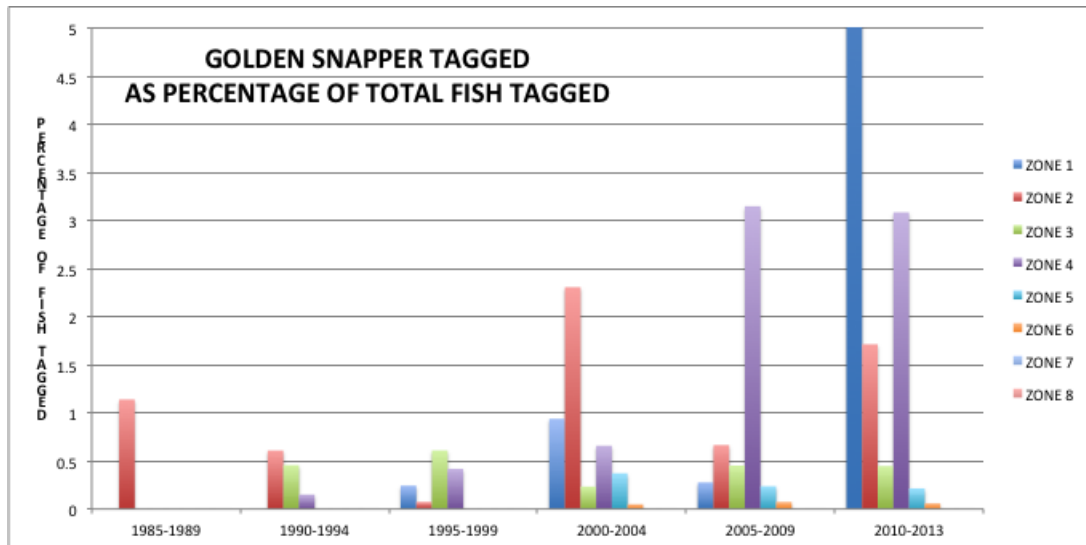
Figure 8.15 Percentage of golden snapper tagged in each zone from 1985-2013
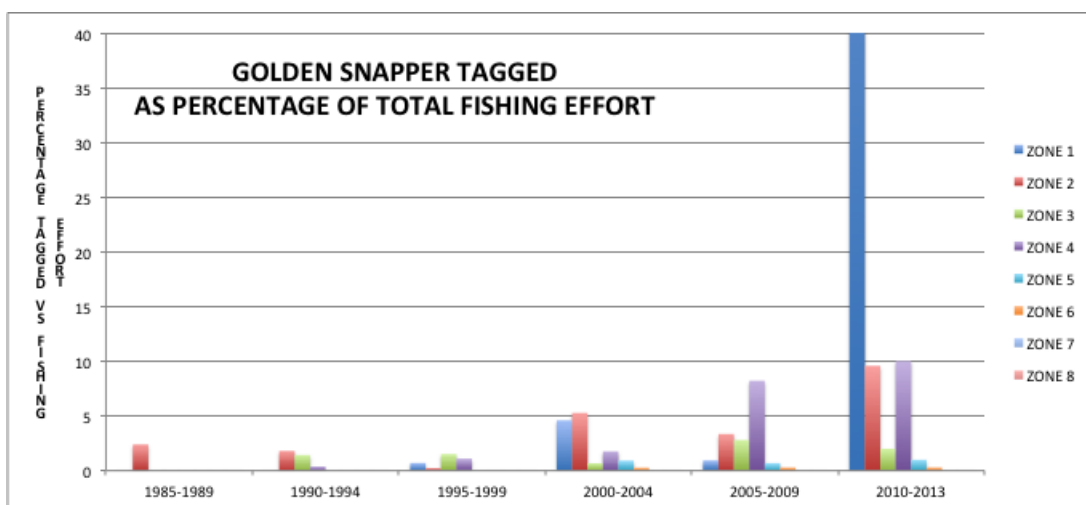
Figure 8.16 Percentage of golden snapper tagged compared with fishing effort in each zone from 1985-2013

From 2000-2012 there were a total of 27 trips to Shoalwater Bay with 10,947 hours of fishing effort and 14,083 fish caught. A total of 1,168 golden snapper were caught on these trips, which was 8.3% of the total fish caught. The area where fish were caught in Shoalwater Bay is in both zones 1 and 2. Figure 8.17 shows the percentage of golden snapper in the catch for each trip and the CPUE through the time series of Shoalwater Bay trips. These data show an increase in the number of golden snapper in the catch, similar to that in the percentage of fish tagged in zones 1 and 2 from normal tagging trips.
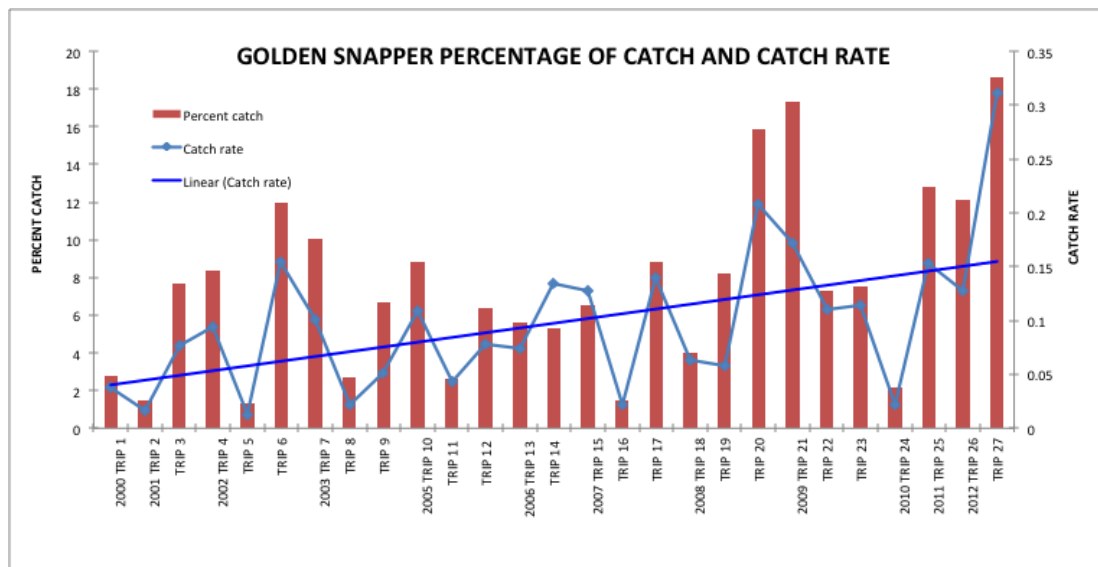
Figure 8.17 Percentage of golden snapper caught and CPUE on Shoalwater Bay trips from 2000-2012.

Of the 766 golden snapper tagged from 1985-2013 there have been 39 recaptures (5.1%), excluding fish from Shoalwater Bay trips. Of these, 35 fish were caught in the same area as tagged or within 10km in the same system. Four fish moved over 20km and were caught outside the system they were tagged in; two fish moved north and 2 fish moved south. A further fish that was presumed tagged back in 1998 (tag data missing) and recaptured over 16 years later in 2013, was also considered to have moved north (Figure 8.18). Of the recaptures, 2 fish were recaptured within approximately 6 months so that the time of movement could be attributed to the Spring/Summer season.

Figure 8.19 shows where Golden Snapper were tagged and recaptured from 1985-2013. The data can be viewed interactively at www.info-fish.net/suntag showing the number of fish tagged in each Suntag grid square over time.

There is anecdotal evidence of golden snapper aggregating in the Port Clinton area in zone 2 during periods when they are known to spawn. There have not been any reports from other areas that suggest spawning sites or aggregations although it is possible that other sites exist.
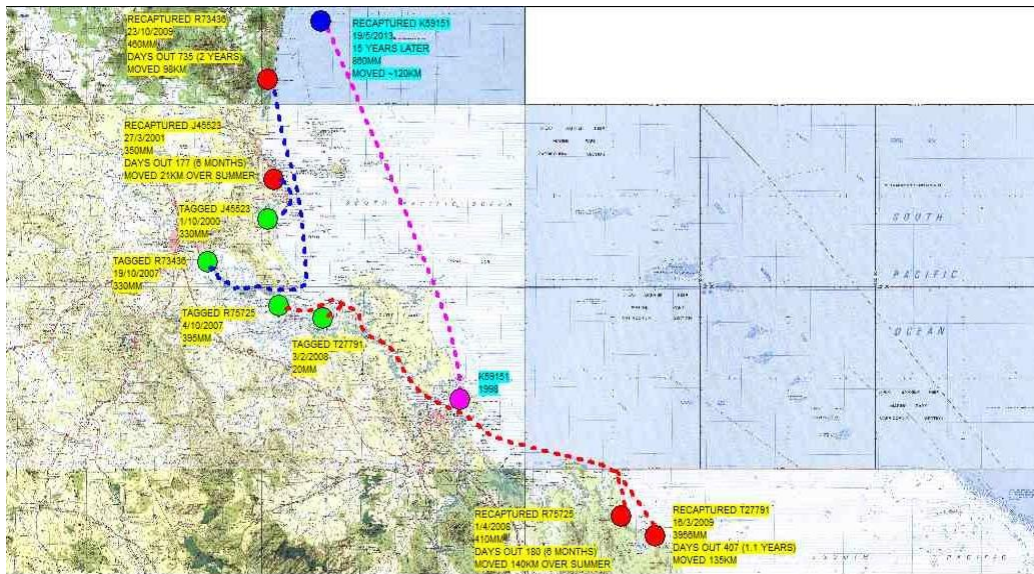
Figure 8.18 Movement of golden snapper tagged in Central Queensland that moved >10 km from the location they were tagged.
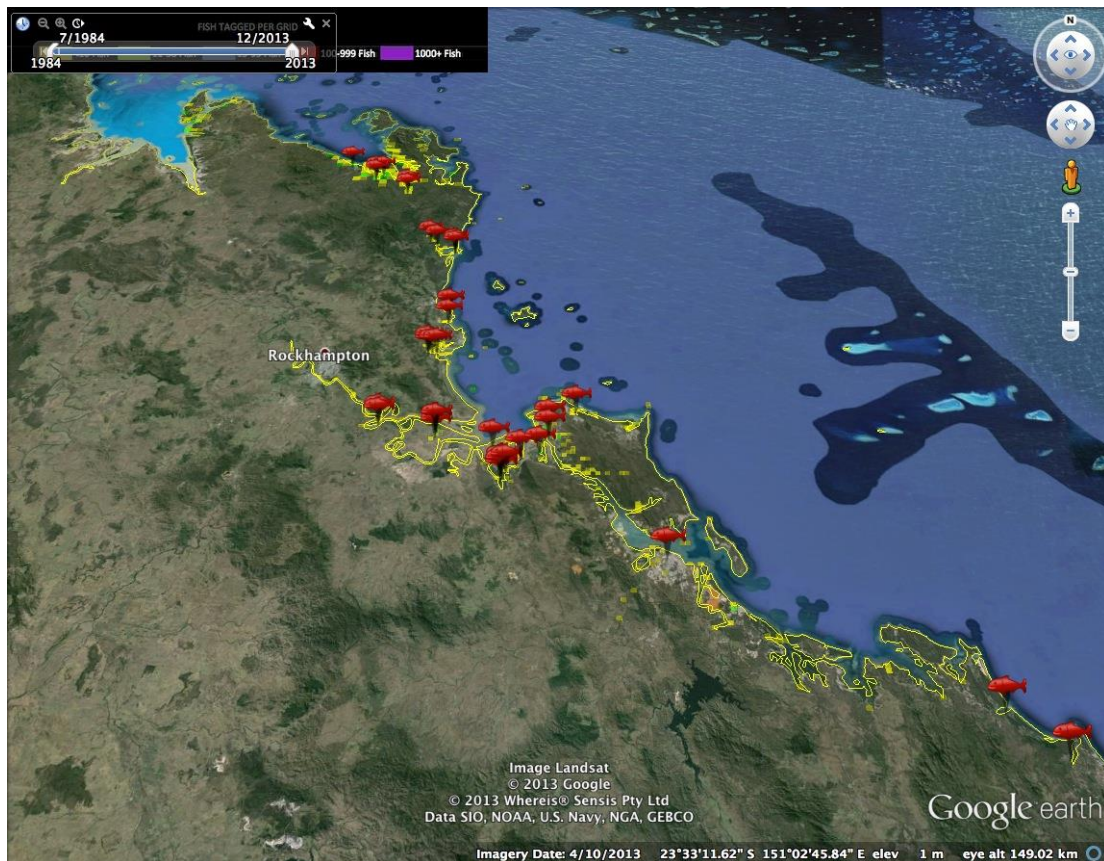
Figure 8.19 Google earth map showing where Golden Snapper were tagged and recaptured from 1985-2013.

Mean SST from 1985-2009 for nearshore areas ranged from 27.6^{o}C at latitude 21.5^{o}S to 25.7^{o}C at latitude 25.5^{o}S in summer, and from 20.9^{o}C at latitude 21.5^{o}S to 21.7^{o}C at latitude 25.5^{o}S in winter. The SST at 25.5^{o}S was the highest winter mean and may be an anomaly due to the inherent difficulties in obtaining accurate satellite-derived SST readings in nearshore areas due to the interference of landmasses (Figure 8.20). There was little or no change in SST at each latitude band over that time (Figure 8.20). However, there was a slight rise in SST of around 0.3^{o}C at latitudes 22.0^{o}S and 23.5^{o}S, but no change was noted at 25.5^{o}S.
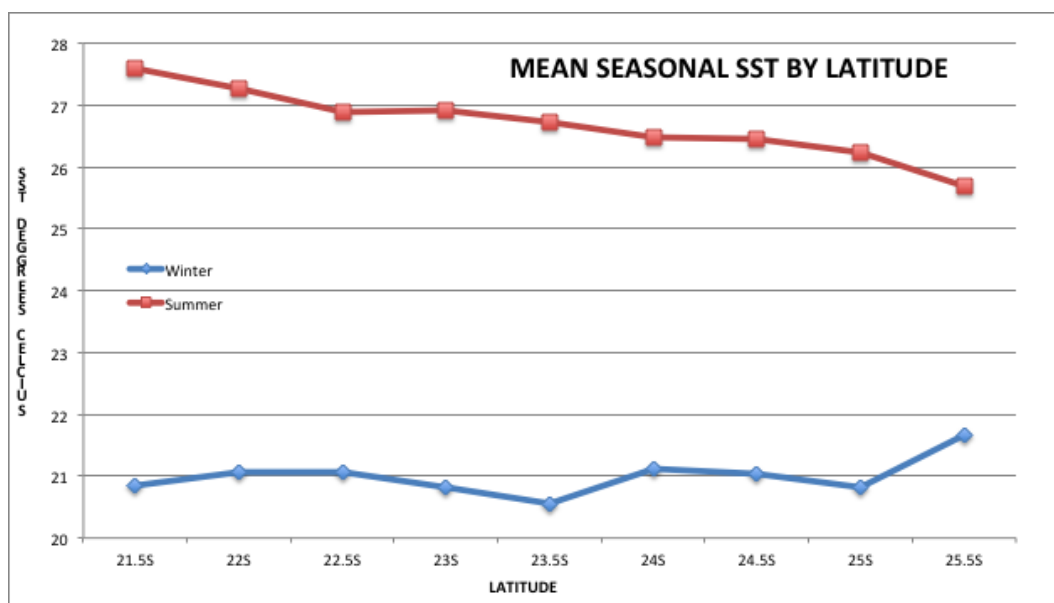
Figure 8.20 Mean summer and winter SST at each latitude from 1985-2009.
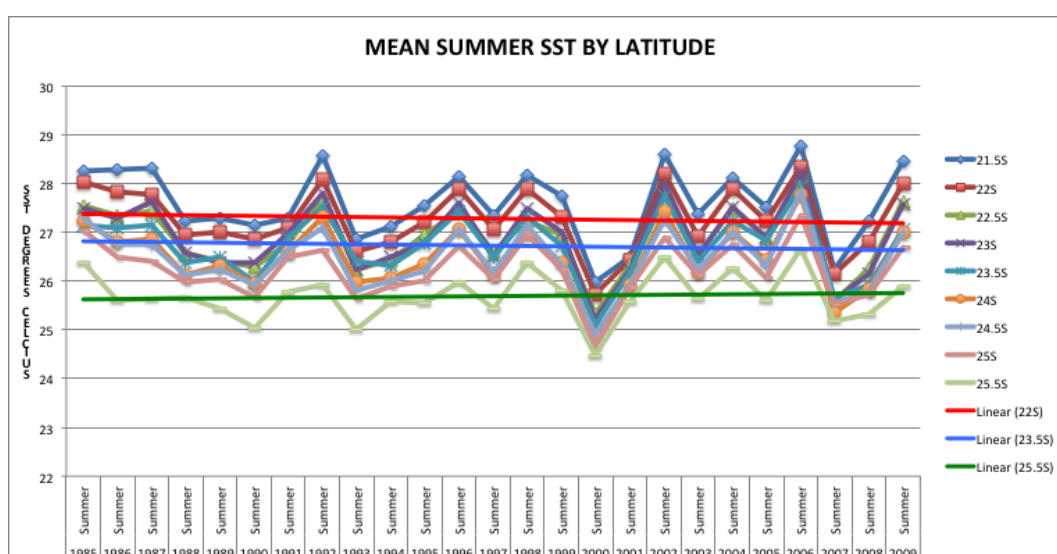
Figure 8.21 Mean summer SST from 21.5^{o}S to 25.5^{o}S from 1985-2009

Figure 8.22 Mean winter SST from 21.5°S to 25.5°S from 1985-2009

8.4.4.2 Discussion

There are no references available for the temperature tolerance range for golden snapper. Therefore it makes it difficult to assess the impact of temperature change on the range of golden snapper. Another possibility is that a reduction in the habitat preferred by golden snapper south of Agnes Water may be a more significant barrier. While there are sufficient suitable estuaries south of Agnes Water the nearshore areas lack rocky headlands and reef that are a preferred habitat of adults. This may limit the ability of golden snapper to increase use of these areas.

There may also be some confusion with identification with Moses Snapper (*Lutjanus russelli*) as both these species are often referred to as Fingermark. This is more likely to occur in the southern zones where there are fewer recordings of golden snapper.

Based on the tagging data presented here, there is evidence that there has been an increase in the numbers of golden snapper that have been caught throughout the time series. This is supported by the separate data collected in the past 12 years of tagging effort in Shoalwater Bay. The exceptions are the two southernmost zones (zones 7 & 8) where there are no records of golden snapper being tagged throughout the time series, and in zone 3 where catches have been stable. Zone 3 represents the area immediately adjacent to the mouth of the Fitzroy River and north to Yeppoon. This area has very high accessibility and is an area where fishing effort has traditionally been high. It is therefore possible that the stable rate of capture of golden snapper in this area, compared with increases in most other areas, is due to prolonged and sustained fishing effort over time. The preferred habitat of golden snapper during the juvenile phase is estuarine and in more nearshore areas as they grow

and mature, but with a preference for turbid waters near headlands and rocky reef structure and with associated sandy areas where they tend to move across and feed. This nearshore turbid habitat tends to change from around the junction of zones 5 & 6 to be clearer water more influenced by oceanic waters, and indeed very few golden snapper were caught in zone 6 during the entire time series and none in zones further south.

It is not possible from the data and these analyses to attribute increased golden snapper captures through time to increasing SST. Based on mean winter SST data there is evidence of slight increases in water temperature over time, which is consistent with broader-scale longer time series trajectories, however there are several other factors that could explain the increased incidence of golden snapper captures. Firstly, targeting of golden snapper is anecdotally known to have increased during this time period. This has led to an increase in the level of skills in catching golden snapper, and there have also been technological advances in GPS and sounders making targeting the fish easier, particularly in turbid nearshore habitats. Indeed, the tagging data since 1990 indicate a general increase in the number of all fish tagged despite tagging effort being relatively stable.

Although there is no clear evidence of a southerly shift in the range of golden snapper, there is an increased rate of capture over time in most of the zones. The recapture data also show fairly clearly that golden snapper do not tend to move very far preferring to be localised in their movements. Even though the data analysed represents a 29-year data set, it is likely that any range shift will take longer to manifest. There is therefore likely to be value in continuing the time series of tagging golden snapper, and other species. If golden snapper do in future years begin to appear in zones 7 and 8, and further south, then this may indeed represent evidence of a range shift, especially given the significant change in habitat types through this area.

8.4.5   Red throat emperor

A close examination of the age frequency distributions by region demonstrated that age at recruitment to the sampling gear differed between the regions (Figure 8.23). Furthermore, the apparent age at recruitment to the sampling gear was not consistent between years, although this is likely to be influenced by low sample sizes in many of the years sampled (Table 8.22). Differences in growth rate do not account for the temporal and regional variation in the age structures or apparent age at recruitment (Williams et al. 2007). Thus, there is evidence to suggest that young red throat emperor recruit to different locations at different ages and that the age at which they arrive at these locations is inconsistent between years. Furthermore, older age classes are not present at Mackay, suggesting either emigration from that location or, as hypothesised by Williams et al. (2007), different rates of mortality exist between regions. These factors indicate that regional treatment of the data to develop year class strength estimates is inappropriate and this should be done using pooled data across all locations.   Pooling data across all regions indicated the appropriate age at recruitment to the sampling gear (standard commercial fishing gear) to be at age 6 (Figure 8.24).
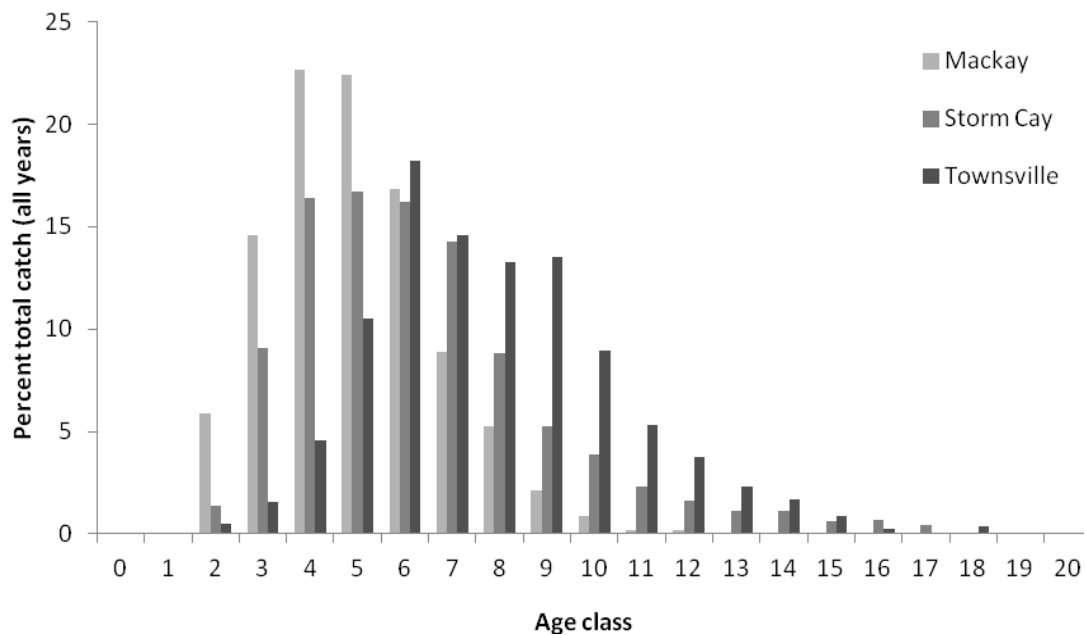
Figure 8.23 Regional age structure of _L. miniatus_ pooled across all years.
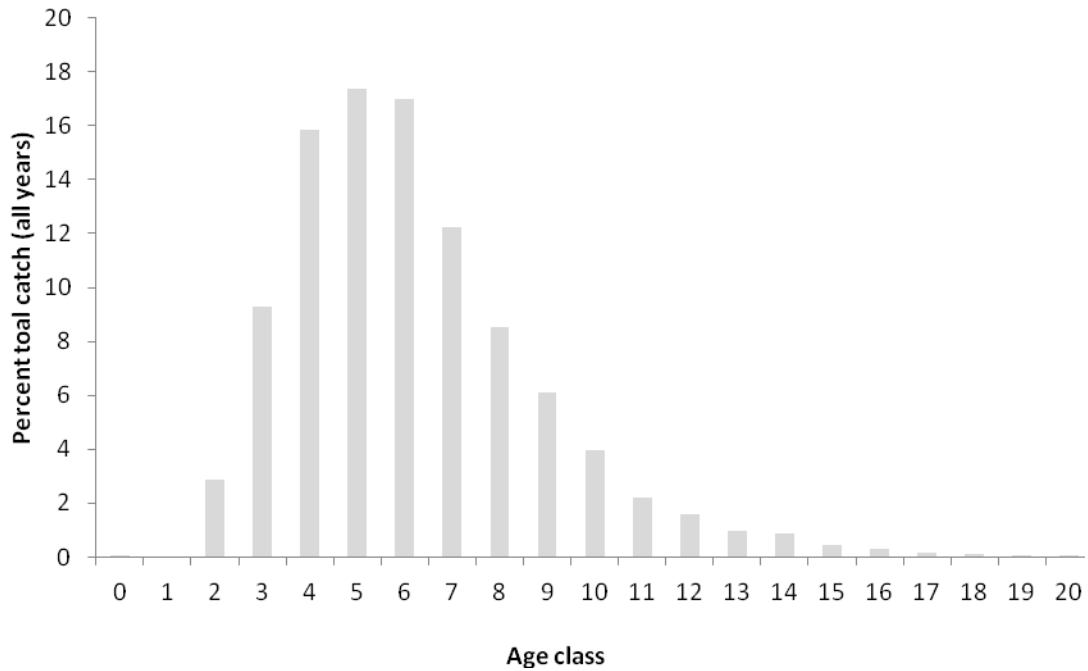
Figure 8.24 Age structure of *L. miniatus* pooled across all regions and all years.

The relative year class strength of red throat emperor over the 11-year period 1995-2005 had the 2003 sampling year excluded from the analyses as it produced a highly aberrant outlier due to the significant underrepresentation of the 1997 cohort. Inter-year variability was significant (ANOVA: *F* = 3.15, *d.f.* =17,206, p < 0.001) implying that recruitment was not constant over time (Figure 8.25). However, it is important to note that while this was significant, it was not a large inter-annual variation in year class strength with the highest and lowest values being approximately +1 and -1.
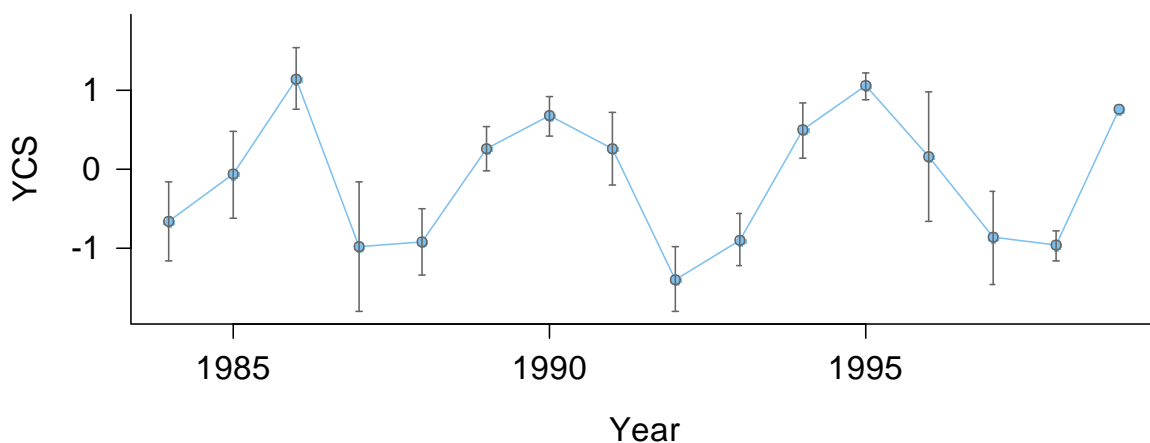
Figure 8.25 Relative year class strength of red throat emperor on the Queensland east coast.

Table 8.22 Population age structure of *L. miniatus* from eastern Australia, 1995-2005, showing numbers of fish sampled per age class.

Region	Age	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Mackay	0	0	0	0	0	0	0	0	0	1	0	0
	2	5	3	9	2	19	25	4	2	3	1	2
	3	11	13	23	18	35	26	20	14	10	6	10
	4	36	8	19	30	66	42	29	15	9	20	15
	5	48	18	6	24	46	28	19	26	3	26	42
	6	39	25	6	4	19	22	22	11	5	14	48
	7	13	25	9	6	4	7	13	11	3	10	12
	8	13	11	2	7	6	5	3	9	2	2	7
	9	6	1	0	3	4	4	0	3	1	1	4
	10	5	2	1	0	0	0	2	0	0	0	1
	11	1	0	0	0	0	0	0	1	0	0	0
	12	0	1	0	0	0	0	0	1	0	0	0
Storm Cay	0	0	0	0	0	0	0	0	0	0	0	0
	2	3	0	1	0	2	5	1	4	0	0	0
	3	12	4	17	16	15	21	4	12	2	1	3
	4	18	4	4	35	63	28	20	5	0	6	11
	5	15	10	11	17	54	38	14	9	1	13	16
	6	39	12	16	8	14	27	22	9	0	11	34
	7	26	17	18	21	15	14	17	12	0	11	18
	8	8	11	9	11	13	17	4	11	0	11	9
	9	14	1	3	7	5	11	2	5	0	5	9
	10	3	7	1	3	5	12	4	1	0	4	6
	11	0	1	1	4	3	4	5	3	0	4	2
	12	0	2	0	6	1	5	2	1	0	0	2
Townsville	0	0	0	0	0	0	0	0	0	0	0	0
	2	1	0	1	0	0	0	0	0	2	0	0
	3	0	0	3	4	0	0	0	1	5	0	0
	4	3	0	7	6	6	3	1	1	1	7	3
	5	23	5	23	4	4	2	0	2	4	6	14
	6	47	12	22	20	10	8	3	1	2	15	11
	7	30	5	16	19	9	7	2	4	9	10	10
	8	27	4	14	18	15	4	1	4	6	13	4
	9	32	3	8	17	12	8	4	1	0	15	12
	10	16	10	10	4	7	6	0	1	4	8	8
	11	5	7	6	8	4	4	1	1	1	4	3
	12	3	2	3	9	2	1	4	2	1	3	1

None of the linear mixed-effect models regressing single environmental variables against year class strength were statistically significant ($p<0.01$) (Table 8.23). While there were no significant correlation found between YCS and the environmental variables measured, the interpretation and analyses were stymied to a large extent as pooling of the age data across a broad geographic range was required. It appears likely that the population of red throat emperor are highly mobile, recruit to different regions/locations at different ages and that these patterns are not consistent between years. Indeed the dramatic low representation of the 1997 year class in the 2003 sampling may not simply be an outlier but an example of part of the population moving out of the sampling area. It is noteworthy in every year some significant recruitment was evident in that there were no missing or extremely low abundance year classes indicating that suitable conditions for recruitment occurred in every year from 1984 to 1999, a period which encompassed a wide range of different environmental conditions.

Table 8.23 Results of mixed effect linear regression of environmental variables against recruitment for *L. miniatus* for each of the three sampled regions within the GBRMP. Offset refers to the lag (-1) or advance (+1) in regressing the time series of environmental variables against the time series of year class strength data. The +/- indicates whether the regression relationship is either a positive or negative one.

Variable	Offset	+/-	F	d.f.	p
SST annual	0	-	0.3187755	1,53	0.5747
	-1	+	4.450333	1,49	0.04
	+1	-	1.8097152	1,56	0.184
SST Spring	0	+	0.7842664	1,53	0.3798
	-1	+	3.273444	1,49	0.0765
	+1	-	0.0939525	1,56	0.7603
SST Spring/Summer	0	+	2.9534492	1,53	0.0915
	-1	+	3.418752	1,49	0.0705
	+1	+	0.1264813	1,56	0.7234
SOI annual	0	+	2.0218331	1,53	0.1609
	-1	+	4.729922	1,49	0.0345
	+1	-	1.0735989	1,56	0.3046
Burdekin flow annual	0	+	1.8064968	1,53	0.1847
	-1	-	3.52253	1,49	0.0665
	+1	+	2.615399	1,56	0.1115
Burdekin flow Wet Season	0	+	2.547322	1,53	0.1164
	-1	+	2.01529	1,49	0.1621
	+1	+	0.3446817	1,56	0.5595

8.4.6 Saucer scallops

8.4.6.1 Characteristics of environmental data

SST fluctuated with season, although there were between year differences in the median SST per grid for any given month (Figure 8.26). Between 1986 and 2001, the greatest anomalies in SST (i.e., difference from overall median for any given month) were +2.03°C (Grid R29, January 1987) and -2.22°C (CFISH Grid R28, December 1999).

Chlorophyll *a* (Chl *a*) fluctuated seasonally, with lowest values occurring most frequently in August and September (Figure 8.27). Highest values of Chl *a* were variable between years in their timing (February through to May) as well as their values (Figure 8.27).

Discharge varied between catchments, but over the study period (1986 to 2012), all of the major rivers adjacent to the Capricorn region experienced large floods as well as periods of extended low flow (Figure 8.28). Of particular note, were the extreme (1-in-100 year) floods in January 2011 that occurred in all rivers of interest. The region has generally low river discharge coincidental to the scallop spawning season (i.e., May to October), with the greatest discharge generally occurring between January and April.

Binary classification of the presence or absence of a cyclonic eddy in the Capricorn region from satellite data provided a simple (although not validated) measure of eddy activity. This data showed considerable variation between years in the eddy activity (i.e., total count) as well as indicated variable timing of eddy presence in relation to the spawning season i.e., early – May to July; or late – August to October (Figure 8.29).
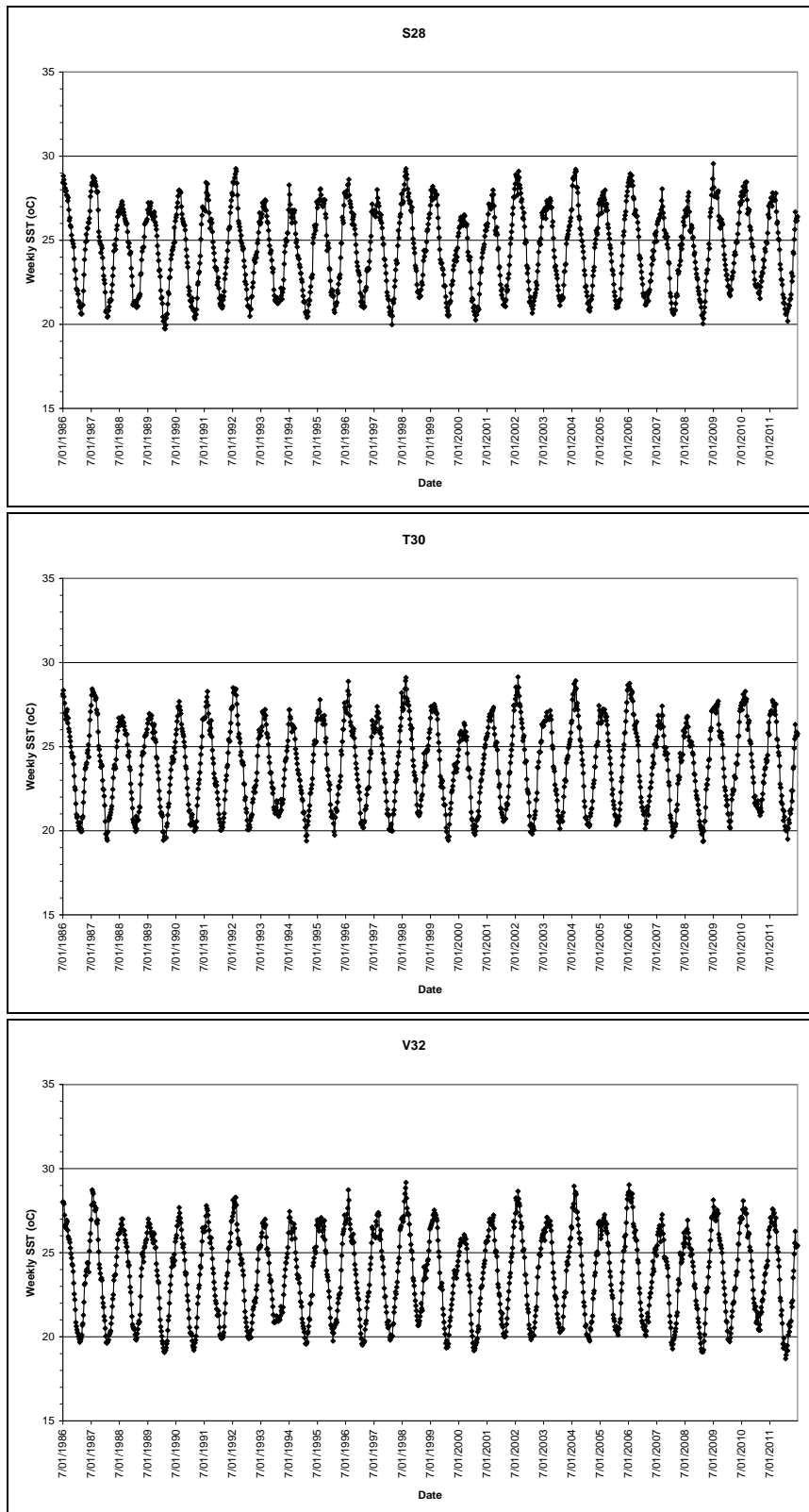
Figure 8.26 Weekly median SST between January 1986 and January 2011 for key scallop (CFISH) grids (S28, T30 and V32) in the Capricorn region of the Queensland east coast.
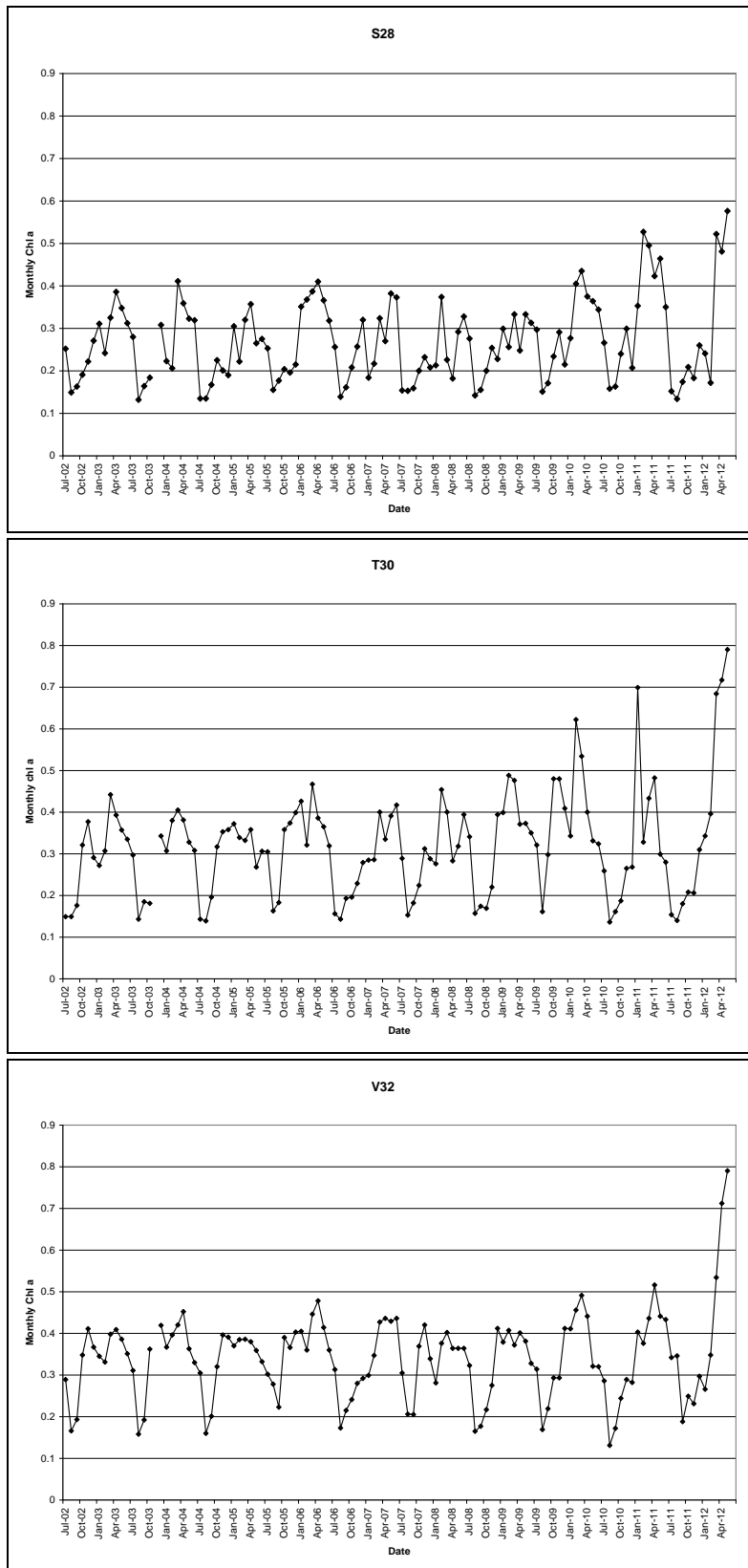
Figure 8.27 Monthly median Chlorophyll a between July 2002 and May 2012 for key scallop (CFISH) grids (S28, T30 and V32) in the Capricorn region of the Queensland east coast.
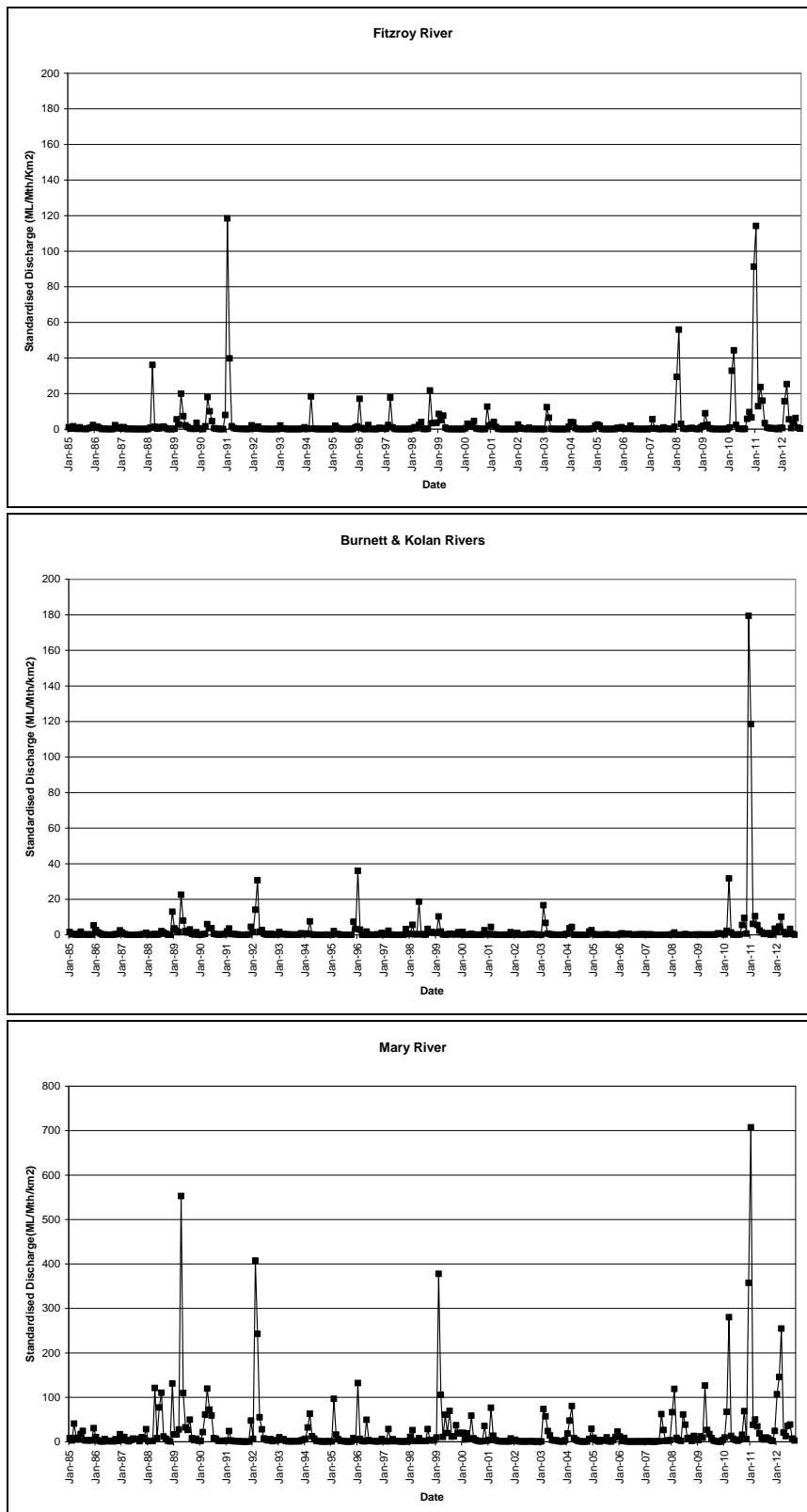
Figure 8.28 Standardised (by catchment area) monthly discharge between January 1986 and September 2012 for rivers influencing the key scallop (CFISH) grids (S28 – Fitzroy; T30 – Burnett & Kolan; and V32 – Mary River) in the Capricorn region of the Queensland east coast.
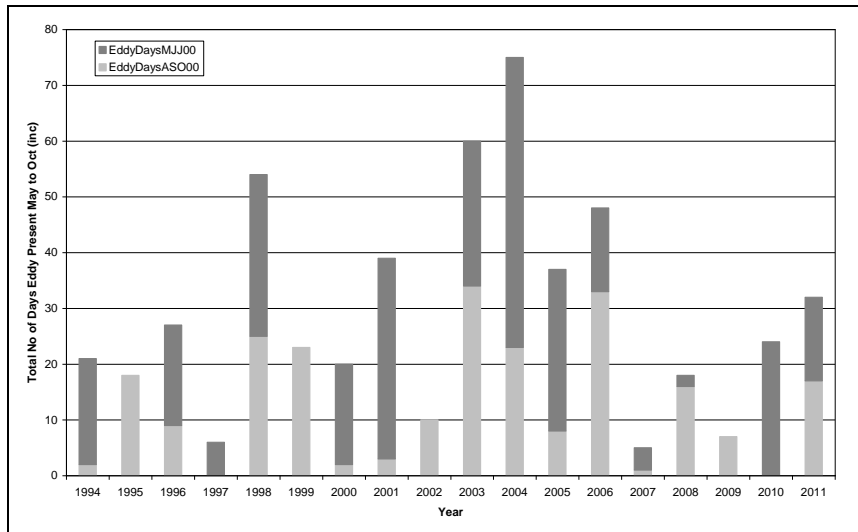
Figure 8.29 Count of the number of days in the scallop spawning season (May to October inclusive) that a cyclonic (hydrological) eddy was visibly present in the Capricorn Region between 1994 and 2011.

Many of the environmental factors in the current study were correlated, with high collinearity between some factors, i.e., r>0.7, as per collinearity diagnostic critical value (Dormann et al. 2012). Collinearity between environmental factors is a known, and almost unavoidable, problem. Some of the observed collinearity was a function of the overlap in the temporal aggregations applied to the data (e.g. Flow Juvenile and Flow Gonads). In other cases, the observed collinearity is probably a function of the interdependence between factors e.g. Chl *a* in Summer is dependent on SST and nutrient input derived from river discharge (Flow). While in other instances, the collinearity probably reflects intra-year climate patterns e.g. SST in Summer, Autumn and Winter.

8.4.6.2 Spatial Recruitment Index

The abundance of 0+ (i.e., recruitment) scallops was highly variable between years and also between cells. The annual commercial catch of scallops per CFISH grid was not significantly correlated to the abundance of 0+ or 1+ scallops. However, the abundance of 1+ scallops was significantly correlated to the abundance of 0+ scallops (r=0.435, p<0.001). Best all sub-sets regression models were used to explore relationships between the abundance of 0+ and 1+ scallops with seasonal Sea Surface Temperature (SST), seasonal and regional river discharge (Flow) and eddy presence (EddyDays). Chlorophyll *a* was not included in the analyses because these data were not available until June 2002, providing an insufficient time series.

Table 8.24 Correlation coefficients (r) environmental factors. Values of r>0.7 (bold type) are indicative of high collinearity between factors.

	Cha Sum	Chla Aut	Chla Win	Chla Spr	SST Sum	SST Aut	SST Win	SST Spr	Flow Gonads	Flow Spawn	Flow Spat	Eddy Days	Eddy Days MJJ	Eddy Days ASO	SOI 6mth Avg	SOI MovAvg06
Chla Sum	1.00															
Chla Aut	0.58	1.00														
Chla Win	-0.11	-0.19	1.00													
Chla Spr	0.16	0.03	0.04	1.00												
SST Sum	**0.72**	0.51	-0.68	0.10	1.00											
SST Aut	0.53	0.37	**-0.83**	0.24	**0.87**	1.00										
SST Win	**0.89**	0.59	-0.15	-0.10	**0.72**	0.57	1.00									
SST Spr	0.00	-0.16	0.58	**0.72**	-0.39	-0.20	-0.09	1.00								
Flow Gonads (Jan to May)	0.44	0.41	-0.59	-0.42	0.48	0.58	0.53	-0.53	1.00							
Flow Spawn (Jun to Oct)	**0.79**	0.58	0.06	-0.29	0.39	0.28	**0.88**	-0.07	0.65	1.00						
Flow Spat (Jul to Nov)	**0.76**	0.60	-0.28	-0.17	0.52	0.53	**0.79**	-0.18	**0.86**	**0.90**	1.00					
Flow Juv (Nov to May)	0.41	0.20	-0.56	-0.38	0.45	0.62	0.57	-0.37	**0.94**	0.64	**0.82**					
EddyDays	0.08	-0.20	**-0.88**	-0.17	0.58	0.69	0.11	-0.58	0.55	-0.08	0.21	1.00				
EddyDays MJJ	-0.09	-0.17	**-0.80**	-0.32	0.36	0.64	0.13	-0.45	0.65	0.04	0.29	**0.86**	1.00			
EddyDays ASO	-0.19	-0.36	-0.48	0.21	0.29	0.20	-0.40	-0.36	-0.23	-0.67	-0.48	0.58	0.16	1.00		
SOI JunNov	0.27	0.09	0.48	-0.55	-0.26	-0.31	0.42	0.04	0.37	**0.75**	0.54	-0.33	-0.07	**-0.80**	1.00	
SOI MovAvg06	0.25	0.16	0.39	-0.59	-0.20	-0.25	0.44	-0.04	0.41	**0.74**	0.54	-0.29	-0.01	**-0.79**	**0.95**	1.00

The base model of "Cell" explained ~32% of variation in the abundance of 0+ scallops (Table 8.25). All sub-sets generalised linear modelling identified several alternate models with adjusted R^2s of between 41 and 52%, depending on the number of terms in the model (i.e., model complexity). Significant factors in the complex models were the Winter sea surface temperature ($p<0.001$) in the same year as the survey (i.e., ~3 months prior), river discharge in the summer to autumn months ($p<0.001$) in the same year as the survey (i.e., ~7 months prior), and eddy presence between May to July ($p<0.001$) in the same year as the survey (i.e., ~ 4 months prior). The parameter estimate for Winter SST indicated an inverse relationship between the abundance of 0+ scallops and Winter SST i.e., 0+ scallops were more abundant when winter sea surface temperatures were lower.

The base model of "Cell" explained ~47% of variation in the abundance of 1+ scallops (Table 8.26). All sub-set generalised linear modelling identified several alternate models with adjusted R^2s of between ~50 and 58%, depending on model complexity. In the most complex 4-term models, the main significant factors were Summer sea surface temperature ($p<0.001$) in the year prior to the survey (i.e., ~20 months before the survey), eddy presence between August and October ($p<0.001$) in the year preceding the survey year (i.e., ~16 months prior), and Autumn sea surface temperature ($p<0.001$) in the year preceding the survey year (i.e., ~18 months prior). The parameter estimates indicated an inverse relationship between the abundance of 1+ scallops and SST in Summer and Autumn in the year preceding the survey i.e., 1+ scallops were more abundant when Summer and Autumn sea surface temperatures were lower when the 1+ scallops were juveniles.

8.4.6.3 Commercial catch data

The commercial catch of saucer scallops has varied dramatically over time (Figure 8.30). Fluctuations in landed catch follow the same trends across the three sub-sets of commercial catch data that were examined. These three sets were: (i) total scallop catch reported within the Queensland East Coast Otter Trawl Fishery (CFISH all data); (ii) scallop catch reported within the Capricorn region (CFISH 22.5° to 26.0° S); and (iii) scallop catch reported by boats selected within the effort standardisation sub-set (Standardisation data). As the trends were similar across data sets, the effort standardisation sub-set was used in further analyses to allow the inclusion of effort creep parameters.

Table 8.25 Best all sub-sets regression models for the abundance of 0+ scallops based on the spatial recruitment index derived by Campbell *et al.* 2011. Base Model = Cell, Adjusted R^2=32.4%.

Adj. R^2	EddyDays MJJ	EddyDays ASO	SST Sum	SST Aut	SST Win	SST Spr	Flow Gonads	Flow Juv	Flow Spat	Flow Spawn	ENSO Class	SOI 6mth Avg
1 terms												
41.3					***							
37.5							***					
36.0								***				
2 terms												
48.8					***		***					
47.4					***			***				
44.5				***	***							
3 terms												
51.2	***				***		***					
49.6	***				***			***				
4 terms												
51.8	***				***		***			*		

[A] Factors in the multiple regression are positively related to the recruitment index unless otherwise indicated in red. *** p < 0.001; ** p < 0.01; * p < 0.05.

Table 8.26 Best all sub-sets regression models for the abundance of 1+ scallops based on the spatial recruitment index derived by Campbell *et al.* 2011. Base Model = Cell, Adjusted R^2=47.3%.

Adj. R^2	EddyDays MJJ01	EddyDays ASO01	SST Sum01	SST Aut01	SST Win01	SST Spr01	Flow Gonads01	Flow Spat01	Flow Juvenile01	ENSO Class01	SOI 6 mth Avg01
1 term											
50.9			***								
49.4					***						
49.2										**	
2 terms											
54.6					***					**	
52.7	**		***								
52.0			***		*						
3 terms											
56.4		***	***	***							
55.9					***			**		**	
4 terms											
57.5	*	***	***	***							
57.1		***	***	***	*						

[A] Factors in the multiple regression are positively related to the recruitment index unless otherwise indicated in red. *** p < 0.001; ** p < 0.01; * p < 0.05.
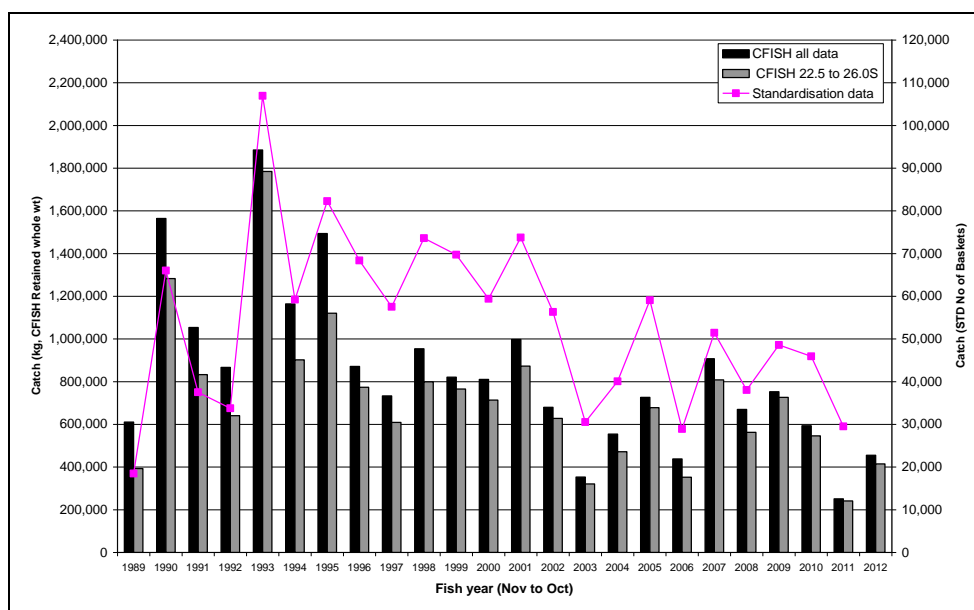
Figure 8.30 Reported catch of scallops in the commercial logbooks of the Queensland east coast otter trawl fishery.

Within the effort standardisation sub-set, total effort (i.e., days of fishing) for scallops increased up to 1997, but then declined (Figure 8.31). Annualised scallop catch per day has fluctuated from about six baskets per boat day (e.g. 1997) to as much as ~20 baskets per boat day (e.g. 1993, 2007, 2009 and 2010).
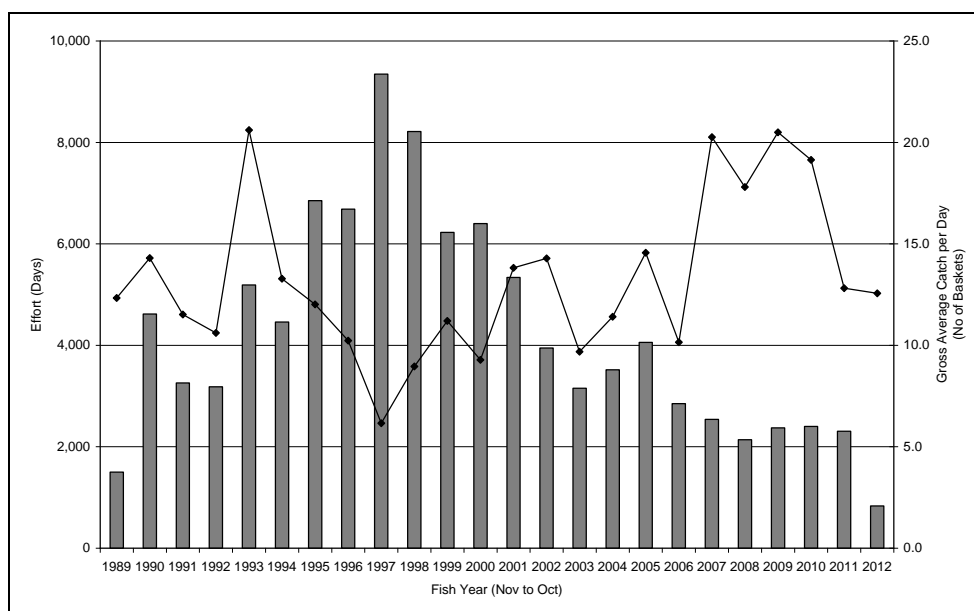
Figure 8.31 Reported effort and gross average catch per day of scallops within the effort standardisation subset of the Queensland east coast otter trawl fishery.

Daily catches per boat were highly variable (average = 16.4 baskets, s.e. = 0.143, min. = 0.1 baskets, max. = 294 baskets); reflecting inter- and intra-year differences in the catch of scallops by different boats. The high variability in daily catch also reflects the large differences in scallop abundance at small spatial scales i.e., within as well as between CFISH grids.

Best all sub-sets regression models were used to explore relationships between the daily catch rates of scallops with seasonal Sea Surface Temperature (SST), seasonal Chlorophyll a, seasonal and regional river discharge (Flow) and eddy presence (EddyDays). Overlap in the time series of catch data with all environmental variables permitted analyses of the data between 2005 and 2011 FishYears inclusive, where 17,728 daily records of catch were available. The forcing month and grid into the model explained ~26.3% of variation in the daily catch of scallops. Significant effort terms of PrawnCatch (-), Lunar(-), LunarAdv, Hrs, Hp, NetType, Gear, Speed (-) and Nozzle were then forced into the model and increased the adjusted R^2 to 44.0%. Best-all subsets identified several alternate complex models that significantly explained variation in daily scallop abundance, with the adjusted R^2 increasing to ~49% (Table 8.27). The environmental terms significantly increased the variation of daily scallop catches explained. However, there remained >50% of the variation in daily scallop

catch rates between 2005 and 2011 that could not be attributed to month, grid, effort terms or environmental terms.

In the most models, the main significant factors were Spring sea surface temperature (p<0.001) in the year immediately prior to harvest (i.e., ≤4 months prior to the majority of the scallop catch in the January and February), and eddy presence between May and July (p<0.001) in the year immediately prior to harvest (i.e., ~7 months prior). Spring SST and EddyDays MJJ accounted for the increase in adjusted R^2, with other significant factors making relatively minor improvements to the adjusted R^2.

The significant models identified for the spatial recruitment index data (i.e., 0+ and 1+ scallop abundance) were also applied to the effort standardisation 2005 to 2011 sub-set (Table 8.27). Although the SRI models held consistent for the commercial catch data, the adjusted R^2 were lower than other models identified by best all-subsets analysis.

Daily catches per boat were highly variable (average = 12.1 baskets, s.e. = 0.0527, min = 0.1 baskets, max = 294 baskets); reflecting inter- and intra-year differences in the catch of scallops by different boats.

Best all sub-sets regression models were used to explore relationships between the daily catch rates of scallops in the effort standardisation sub-set with seasonal SST, seasonal and regional Flow and EddyDays i.e., seasonal Chlorophyll *a* was excluded. Overlap in the time series of catch data with these environmental factors permitted analysis of the data between 1996 and 2011 FishYears inclusive, providing 63,142 daily catch records. Forcing month and grid into the model explained ~18.2% of variation in the daily catch of scallops. Significant effort factors of PrawnCatch (-), Lunar(-), LunarAdv, Hrs, Hp, NetType, Gear, Speed (-) and Nozzle were then forced into the model and increased the adjusted R^2 to 37.8% (Table 8.28).

Best-all subsets identified several alternate complex models that significantly explained variation in daily scallop abundance, with the adjusted R^2 around 40% (Table 8.28). Whilst environmental factors significantly increased the adjusted R^2, there remained >60% of the variation in daily scallop catch rates between 1996 and 2011 that could not be attributed to month, grid, effort or environmental factors.

The significant models identified for the spatial recruitment index data (i.e., 0+ and 1+ scallop abundance) were applied to the effort standardisation 1996 to 2011 sub-set (Table 8.28).

Table 8.27 Best all sub-set regression models for the daily catch of scallops reported in the commercial logbooks of the Queensland east coast otter trawl fishery – effort standardisation sub-set for the Capricorn region 2005 to 2011 FishYears (incl. Chl-a data). Base model = Month + Grid + Effort Factors, Adj. R^2 = 44.0.

Adj. R^2	EddyDays MJJ	EddyDays ASO	EddyDays MJJ01	EddyDays ASO01	SST Sum01	SST Aut01	SST Win01	SST Spr01	SST Sum	SST Aut	SST Win	SST Spr	Chla Sum01	Chla Aut01	Chla Win01	Chla Spr01	Chla Sum	Chla Aut	ChlA Win	Chla Spr	Flow
1 terms																					
46.0					***																
45.7																					*** Spawn
45.6												***									
2 terms																					
48.5	***											***									
47.6										***		***									
47.2				***												***					
3 terms																					
48.8	***	***										***									
48.8	***											***			***						
48.7	***											***				***					
4 terms																					
49.2	***											***			***			***			
49.2	***										***	***			***						
49.1	***			***								***				***					
5 terms																					
49.5	***			***		***						***				***					
49.4	***			***		***						***			***						
49.3	***										***	***			***	***					

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SRI 0+ model	1	2	3	4	5	6	7	8	9	10	11	12							
46.7	***										***								*** Gonads
SRI 1+ models																		
46.7					***					***									
47.1	***				***	***													
SRI 0+ and 1+ combined																		
47.3	***				***	***					**								*** Gonads
46.2					***						***								
47.1	***				***	***													
46.6					***	***					***								
47.2	***				***	***													*** Gonads

Factors in the multiple regression are positively related to the standardised scallop catch unless otherwise indicated in red. *** p < 0.001; ** p < 0.01; * p < 0.05.

Table 8.28 Best all sub-set regression models for the daily catch of scallops reported in the commercial logbooks of the Queensland east coast otter trawl fishery – effort standardisation sub-set for the Capricorn region 1996 to 2011 FishYears (No Chl-a data). Base model = Month + Grid + Effort Factors; Adj. R^2 = 37.8.

Adj. R^2	EddyDays MJJ	EddyDays ASO	EddyDays MJJ01	EddyDays ASO01	SSTS um01	SST Aut01	SST Win01	SST Spr01	SST Sum	SST Aut	SST Win	SST Spr	Flow Gonads
1 term													
38.5												***	
38.4									***				
38.3						***							
2 terms													
39.1									***			***	
39.0											***	***	
38.9		***							***				
3 terms													
39.5								***	***			***	
39.5						***		***	***				
39.4		***							***			***	
4 terms													
40.1						***		***			***	***	
39.9						***		***	***			***	
39.8					***			***	***			***	
5 terms													

40.5				***		***	***		***	***	
40.2	***			***		***			***	***	
40.2		***				***	***		***	***	
40.2			***			***	***		***	***	
40.1				***	***	***			***	***	
SRI 0+ model											
38.1	***								***		***
SRI 1+ models											
38.3				***	***			**			
38.3	***			***	***						
SRI 0+ and 1+ combined											
38.6	***			***	***				***		*
38.2				***			***				***

Factors in the multiple regressions are positively related to the standardised scallop catch unless otherwise indicated in red. *** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$.

8.4.6.4 Discussion

Worldwide, scallop fisheries are notorious for their large fluctuations in abundance and subsequent variability in catch. Queensland saucer scallops are no exception.

Analysis of the fishery-independent recruitment survey data (i.e., spatial recruitment index) indicated that SST may play a significant role in influencing Queensland scallops during their larval and juvenile phases, with cooler temperatures have a positive influence. This fits with the known literature for Queensland saucer scallops, where a key trigger for spawning under laboratory conditions is falling water temperatures. The presence of a cyclonic eddy in the Capricorn region over of the spawning season was indicated in the analysis to have a positive effect on scallop abundance, although results were somewhat ambiguous as to the importance of timing. The significant and positive effect of river discharge on the abundance of 0+ scallops in the seven months prior to the survey may reflect enhanced nutrient conditions leading to better growth and survival of young-of-the-year scallops.

Analysis of the fishery-dependent daily commercial catch data provided ambiguous results. Despite finding many significant environmental effects, these ultimately only increased the amount of variation explained by a small amount. Many issues complicated the commercial catch data including major management changes, the reliability of reported data, and its low spatial resolution (i.e., CFISH grid = ~167 km^2). As a coarse generalisation, sea surface temperature was a consistently significant factor in many of the GLMs for the daily commercial catch, but its seasonality varied between models making interpretation difficult.

Despite significant results for the fishery-independent scallop recruitment survey data, >50% of the variation in the abundance of 0+ and >60% for 1+ scallops remained unexplained by environmental factors. Results were similar for the more complex commercial catch data. This indicates that there are probably other factors influencing scallop abundance. Some are known, such as the spatial and temporal closures used as part of the management regime for Queensland saucer scallops, but which were difficult to include in the present analysis. It is highly likely factors occurring at very small spatial scales (e.g. <1km^2) play a significant role in determining the abundance of Queensland saucer scallops. Such factors could include bottom composition, bottom aspect, or exposure to tidal currents. Queensland saucer scallops have a known preference for substrates that are soft (to enable the scallops to burrow), and with a high sand content. Survey results indicate that substrate composition can vary at quite small spatial scales (Pitcher et al. 2007).

Queensland saucer scallops occur in a relatively narrow area on the Queensland east coast between 22°30' S, 151° E and 26° S, 153°30' E. Occasionally, significant patches of scallops are reported north (up to Hydrographers Passage) and south (in front of Fraser Island) of this key area. Their current distribution indicates that saucer scallops are spatially

constrained, probably by several environmental factors in addition to depth. The results from the current analysis indicate that water temperature is probably a key factor. This is consistent with the observed effects of a Marine Heat Wave in Shark Bay Western Australia where water temperatures >3°C above average resulted in reduced growth rates of saucer scallops and recruitment failure (Pearce *et al*. 2011). The WA experience suggests that extreme events (particularly high temperature combined with lowered salinity i.e., a flood) had major negative effects on saucer scallops. It is unknown whether such extreme events can occur in the Capricorn region, given its differing geographic and hydrographic conditions to that of Shark Bay in Western Australia.

Results were somewhat ambiguous as the effect on saucer scallops of cyclonic eddies derived from the East Australian Current. This is probably the consequence of the simplistic index of eddy presence generated in the current study. Further work on understanding the (probably complex) influence of large scale hydrographic features on larval dispersal and spatfall would assist our understanding of potential changes to saucer scallops that may result from a changed East Australian Current.

8.4.7 Spanish mackerel

8.4.7.1 Year class strength

Monitoring of the length- and age-structure of east coast Spanish mackerel occurred between 2001 and 2011. Age-length keys derived from 8270 aged fish were used to assign ages to a further 22145 measured fish resulting in a total sample of 30415 Spanish mackerel (see Appendix 10). Full selectivity to the fishing gear and recruitment to the line fishery appeared to occur at age two, and fish >11 years were rare. Consequently fish 2–11 years old were included in the analysis. Based on catch-curve residuals it was possible to reconstruct the relative YCS of Spanish mackerel over the 16-year period from 1993-2009.

Comparing time-series of mean YCS from four different regions along the east coast of Townsville indicated that trends in recruitment were homogenous across the study area (Table 8.29, Figure 8.32). All regions were highly significantly correlated. Townsville and Mackay showed the least amount of similarity (0.78) while the greatest similarity was between Townsville and South (0.96). Over the range of years available there appeared to be evidence of one particularly strong year of recruitment in 2007, and two relatively weak periods of recruitment; 1998-1999 and 2004-2006. The r^2 values from catch curve linear regression models that included age, x, and sample year, i, ranged from 0.71 from Rockhampton (most variability in recruitment) to 0.87 for Mackay (least variability in recruitment). The Townsville and South regions were relatively similar with r^2 values of 0.80 and 0.82, respectively.
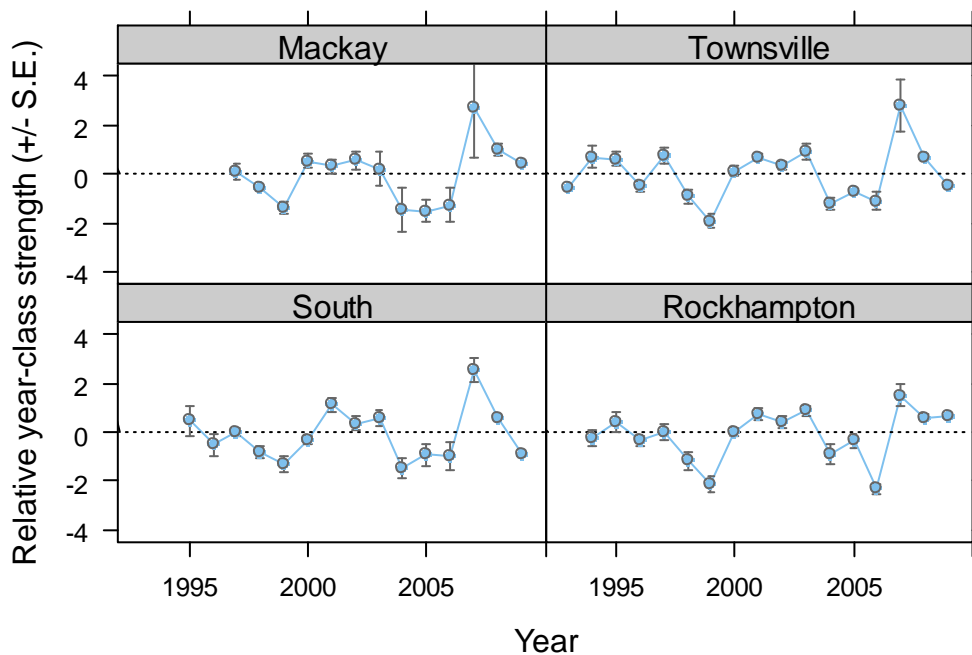
Figure 8.32 Relative year-class-strength of Spanish mackerel in four regions on the east coast of Queensland.

Table 8.29 Pearson's correlation coefficient between indices of year-class-strength of Spanish mackerel in four geographical regions. Values > 0.76 are significant at the 0.005 level.

	Townsville	Mackay	Rockhampton	South
Townsville				
Mackay	0.78			
Rockhampton	0.89	0.82		
South	0.96	0.86	0.91	

8.4.7.2 Catch per unit effort

All fitted terms included in the linear mixed model for standardising CPUE were highly significant (Table 8.30). Despite this, there was relatively little difference between the fitted year coefficients and the geometric mean daily catch rates. CPUE initially showed a declining trend however from the mid-1990s but increased steadily with a noticeable peak in 2009 (Figure 8.33).
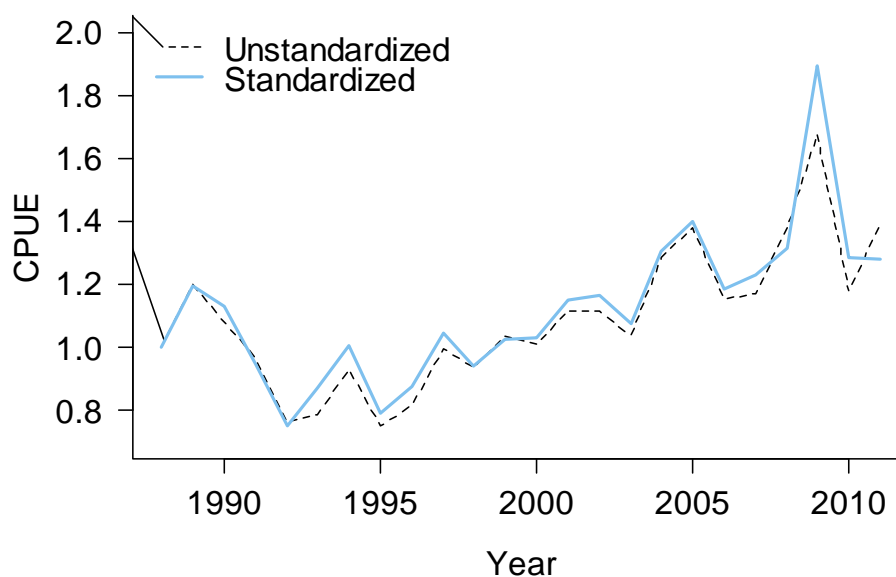
Figure 8.33 Comparison of standardised CPUE and geometric mean of daily unstandardised catch rates relative to 1988 levels.

Table 8.30 Wald tests showing statistical significance of fitted terms in the linear mixed model used to standardise CPUE.

Fitted terms	Numerator d.f.	Denominator d.f.	F	P
Intercept	1	27270	5064.27	<0.0001
Year	23	27270	29.444	<0.0001
Month	11	27270	136.213	<0.0001
Lunar (sine)	1	27270	273.971	<0.0001
Lunar advance (cosine)	1	27270	161.802	<0.0001
Lunar (sine 2)	1	27270	23.168	<0.0001
Lunar advance (cosine 2)	1	27270	63.44	<0.0001
Grid	8	27270	20.965	<0.0001

8.4.7.3 Comparison of abundance indices

Pearson's correlation coefficient between CPUE and the YCS (pooled across regions) from two years prior was 0.52 indicating a significant positive correlation between the two indices (t = 2.3729, *d.f.* = 15, p = 0.03). This correlation was not particularly robust, however, and appeared mainly to be driven by data from the period between 2004 and 2011 (Figure 8.34). During this time the period of relatively weak recruitment (2004-2006) and the strong year of recruitment (2007) were both evident in the CPUE. After removal of the strong recruitment year the correlation was no longer significant.
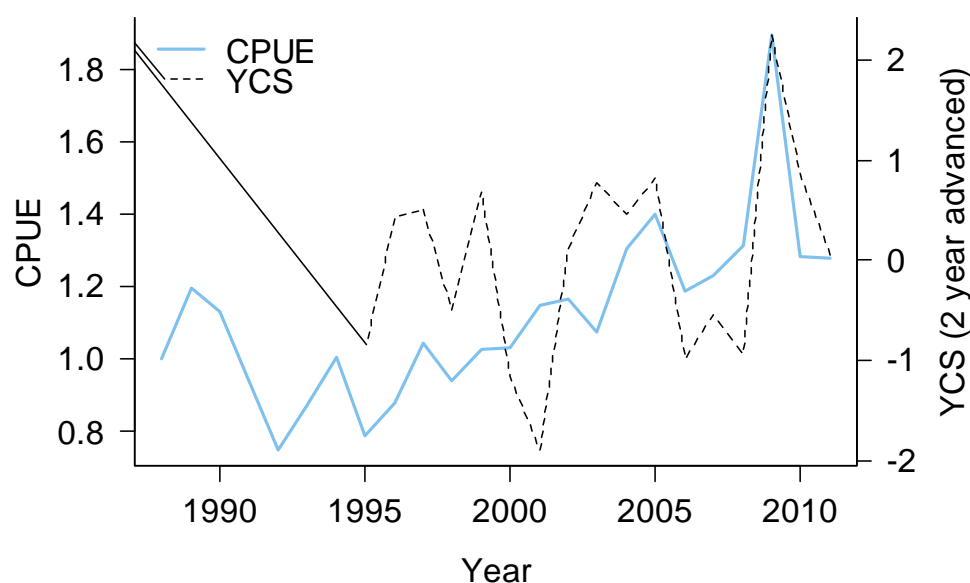
Figure 8.34 Comparison of CPUE with the YCS advanced two years, corresponding with the lag between spawning and full recruitment to the fishery. Both indices are dimensionless.

8.4.7.4 Regression analysis — Year class strength

A total of 48 bivariate linear regression models correlating environmental variables against YCS residuals were analysed, six of which were statistically significant ($P < 0.01$) (Table 8.31). SST was the only variable that had a strong and consistent correlation with YCS; in all regions there was a strong negative and one year lagged response of Spanish mackerel YCS with Spring SST. The greatest statistical significance was in the Townsville and Rockhampton regions where regressions explained 35% and 44% of the remaining variability in YCS not accounted for in the base model, respectively (Figure 8.35, Figure 8.36). The strong negative lagged correlation in the Townsville and Rockhampton region also manifested as a year advanced positive correlation that was statistically significant. The same regressions in the Mackay and South region were still statistically significant however only explained 18% of the remaining variability.
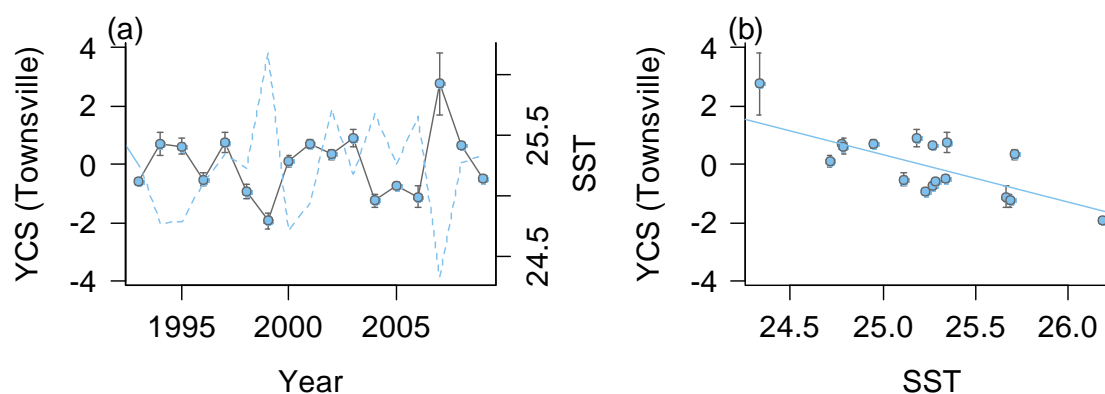
Figure 8.35 Comparison of mean Spanish mackerel YCS against lagged Spring SST (dashed blue line) (a), and YCS as a function of lagged SST in the Townsville region (b). The solid blue line in (b) is a linear regression between the two variables (Table 8.31).

Other environmental variables had a less consistent response. There was a strong positive lagged response of YCS to Spring Chl-a, however only in the Townsville region. One year advanced SOI had a strong positive correlation in the Rockhampton region. A statistically significant but weaker negative correlation occurred with no lag.

Table 8.31 Results of linear regression of single environmental variables against catch-curve residuals. Models that were statistically significant at p ≤0.01 are denoted in bold.

Variable	Offset	Region	+/-	F	d.f.	P	r^2	Region	+/-	F	d.f.	P	r^2
SST	0	Townsville	+	2.33	1,96	0.13	0.02	Mackay	+	0.33	1,52	0.57	0.01
	-1		**-**	**52.79**	**1,96**	**<0.01**	**0.35**		**-**	**11.66**	**1,52**	**<0.01**	**0.18**
	+1		**+**	**12.58**	**1,95**	**<0.01**	**0.12**		+	3.92	1,51	0.05	0.05
SOI	0		-	4.98	1,96	0.03	0.05		+	0.67	1,52	0.42	0.01
	-1		-	4.40	1,96	0.04	0.04		+	0.00	1,52	0.96	0.00
	+1		-	1.80	1,96	0.18	0.02		-	0.10	1,52	0.76	0.00
Chl-a	0		-	0.19	1,34	0.66	0.01		-	1.84	1,30	0.19	0.06
	-1		**+**	**52.99**	**1,26**	**<0.01**	**0.67**		-	2.19	1,23	0.15	0.09
	+1		+	0.58	1,43	0.45	0.01		+	0.58	1,36	0.45	0.02
River flow	0		-	5.03	1,96	0.03	0.05		+	1.89	1,52	0.04	0.18
	-1		-	5.12	1,96	0.03	0.05		-	5.20	1,52	0.03	0.09
	+1		-	3.31	1,96	0.07	0.03		-	0.19	1,52	0.66	0.00
SST	0	Rockhampton	+	1.71	1,81	0.19	0.02	South	+	0.00	1,67	0.96	0.00
	-1		**-**	**64.77**	**1,81**	**<0.01**	**0.44**		**-**	**14.27**	**1,67**	**<0.01**	**0.18**
	+1		**+**	**6.61**	**1,80**	**0.01**	**0.08**		+	4.41	1,66	0.04	0.06
SOI	0		-	1.23	1,81	0.27	0.01		-	0.06	1,67	0.80	0.00
	-1		-	3.71	1,81	0.06	0.04		-	0.62	1,67	0.44	0.01
	+1		**-**	**11.04**	**1,81**	**<0.01**	**0.12**		-	4.69	1,67	0.03	0.07
Chl-a	0		-	0.77	1,34	0.39	0.02		+	5.43	1,34	0.03	0.14
	-1		+	0.16	1,26	0.69	0.01		-	0.01	1,26	0.94	0.00
	+1		+	1.44	1,43	0.24	0.03		-	0.21	1,42	0.65	0.00
River flow	0		+	2.96	1,81	0.04	0.09		-	3.21	1,67	0.08	0.05
	-1		-	0.15	1,81	0.70	0.00		-	2.81	1,67	0.10	0.04
	+1		+	2.23	1,81	0.14	0.03		-	0.16	1,67	0.69	0.00
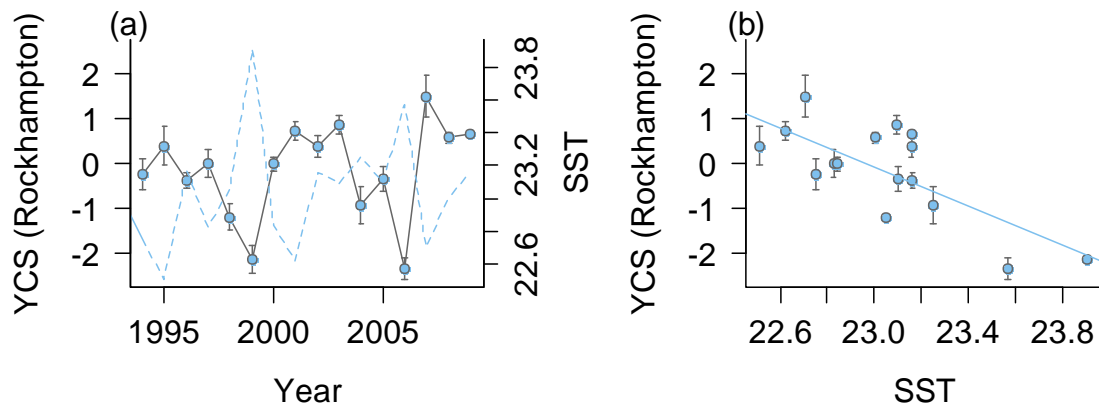
Figure 8.36 Comparison of mean Spanish mackerel YCS against lagged Spring SST (dashed blue line) (a), and YCS as a function of lagged SST in the Rockhampton region (b). The solid blue line in (b) is a linear regression between the two variables (Table 8.31).
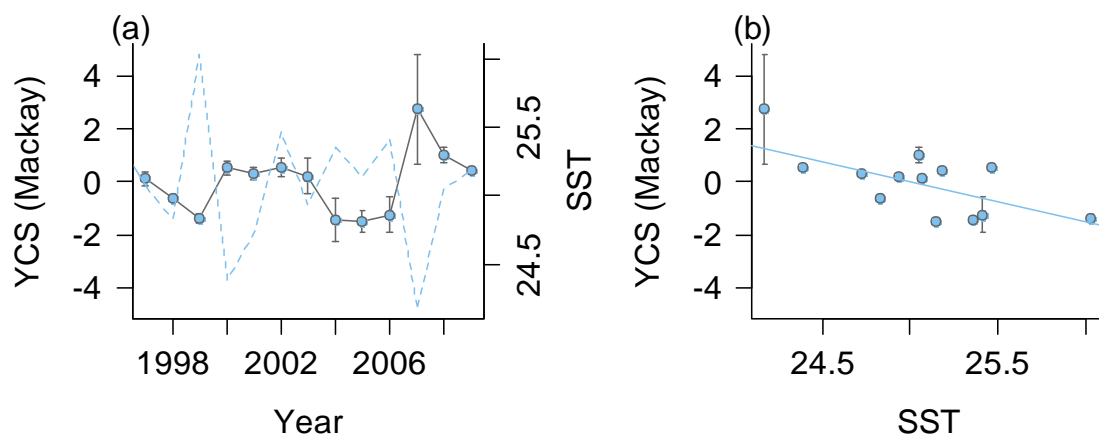
Figure 8.37 Comparison of mean Spanish mackerel YCS against lagged Spring SST (dashed blue line) (a), and YCS as a function of lagged SST in the Mackay region (b). The solid blue line in (b) is a linear regression between the two variables (Table 8.31).

8.4.7.5 Regression analysis — Catch per unit effort

A total of 12 bivariate linear regression models correlating environmental variables against standardised annual CPUE were analysed, 3 of which were significant at the 0.05 level. Both SOI and CPUE increase at a similar rate from 1993 to 2011 and the most statistically significant relationship was a positive correlation between lagged SOI and CPUE. Lagged river flow, which was correlated with lagged SOI had a weak but statistically significant correlation with CPUE. One year advanced SOI was also weakly statistically significant.
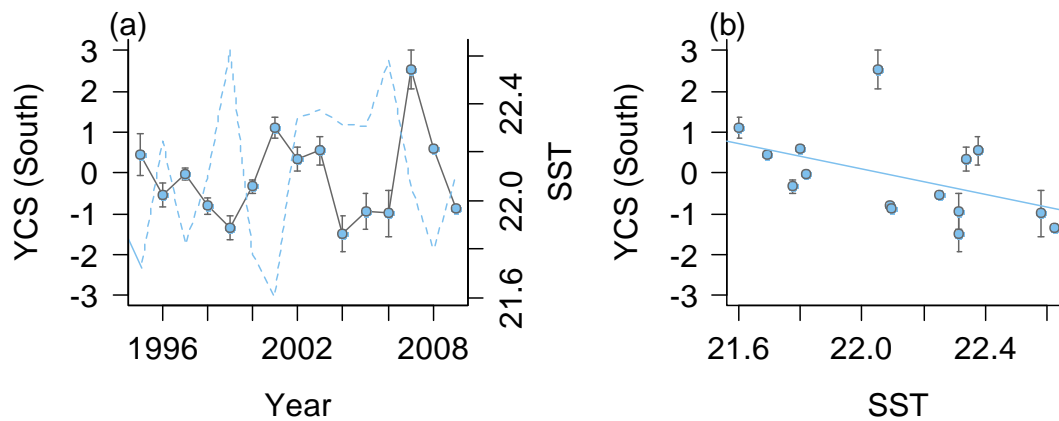
Figure 8.38 Comparison of mean Spanish mackerel YCS against lagged Spring SST (dashed blue line) (a), and YCS as a function of lagged SST in the South region (b). The solid blue line in (b) is a linear regression between the two variables (Table 8.31).

Table 8.32 Results of linear regression of single environmental variables against CPUE in the Townsville region. Models that were statistically significant at p ≤0.05 are denoted in bold.

Variable	Offset	+/-	F	d.f.	P	r^2
SST	0	+	1.22	1,20	0.28	0.06
	-1	+	1.15	1,21	0.30	0.05
	+1	+	0.21	1,19	0.65	0.01
SOI	0	+	0.24	1,22	0.63	0.01
	-1	+	**7.74**	**1,22**	**0.01**	**0.26**
	+1	+	**4.64**	**1,21**	**0.04**	**0.18**
Chl-a	0	+	0.20	1,8	0.67	0.02
	-1	+	0.00	1,7	0.96	0.00
	+1	+	0.18	1,8	0.68	0.02
River flow	0	+	0.97	1,22	0.34	0.04
	-1	+	**4.81**	**1,22**	**0.04**	**0.18**
	+1	+	3.29	1,21	0.08	0.14
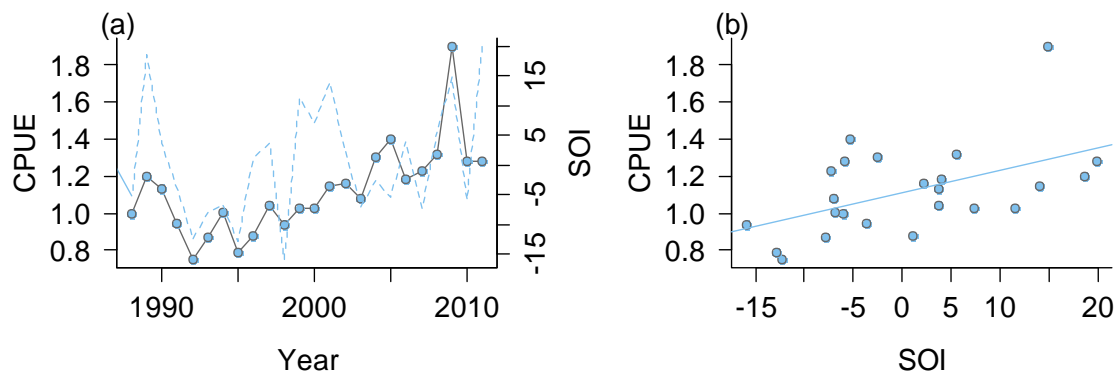
Figure 8.39 Comparison of standardised annual CPUE against lagged SOI (dashed blue line) (a), and CPUE as a function of lagged SOI in the Townsville region (b). The solid blue line in (b) is a linear regression between the two variables (Table 8.32).

8.4.7.6 Discussion

We tested four hypotheses relating environmental variables with a catch-curve based YCS index and a CPUE based abundance index for *S. commerson* on the east coast of Queensland. We found that 71-87% of variation in YCS in four broad-scale regions on the east coast of Queensland was explained by age and sample year, suggestive of moderately variable recruitment in this species. Additionally, trends in recruitment were found to be highly consistent throughout the study region with values of Pearson's correlation coefficient between all four YCS regions ranging from 0.78 to 0.96. One-year lagged Spring SST was found to a have a strong negative correlation with YCS, explaining up to 44% of the remaining variation not accounted for in the base models. Although this correlation was consistent across all four regions where YCS was measured, it was not consistent with the original hypotheses that suggested SST at the time of spawning itself and during the period of larval development (rather than one year lagged) would affect recruitment. As such, we could not immediately establish a causal mechanism to explain this correlation. A strong positive correlation was also found between standardised CPUE and one-year lagged SOI, however this was also inconsistent with our hypothesis, and a causal mechanism was not clear. There was a weakly significant positive correlation between YCS and CPUE indices at the expected lag of two years (age at full recruitment to the fishery), providing some indication that the effects of strong and weak year classes influence catch rates in this fishery. This has been assumed in the past but until now had not been tested.

Sea surface temperature

The negative and one-year lagged correlation between spring SST and *S. commerson* YCS is an intriguing but unexpected finding from this study. Review of the lifecycle of *S. commerson* indicated that spring SST could potentially be a key environmental variable, affecting recruitment by influencing the timing of spawning, egg production and larval

survival, and potentially affecting growth and catchability. The one-year lag between the correlation, however, suggests an indirect effect. Furthermore, the correlation was in the opposite direction to what was expected, and implied that increasing temperature has a negative effect on *S. commerson*. While a negative response to higher temperature is not uncommon in itself, this would typically be expected to occur at the lower-most latitude that the species occurs in (Myers 1998). One possibility is that juvenile *S. commerson* survival is enhanced by higher baitfish recruitment from the previous year. Townsville is known as the main spawning ground, and therefore the main source of recruits, for the east coast. Cross-shelf incursions of cooler upwelled water, typically nutrient-rich, have been reported for the Townsville region and there was a significant *positive* and one-year lagged correlation between Chlorophyll-a. and *S. commerson* YCS in the Townsville region only. This could promote increases in baitfish populations that juvenile mackerel take advantage of in the first 6 months of life. If so, this would suggest that survival at the juvenile stage, rather than the larval stage, is more critical for recruitment to the fishery population. This is a possible indirect response that would explain the results here, however is a new hypothesis that would require testing. Also, as pointed out already, it would be assumed that higher nutrients in the year of spawning and larval development would enhance survival through greater food availability. This was not observed from the analyses. It is interesting to note that during the 1970s, fishers targeted spawning aggregations of *S. commerson* on reefs between Townsville and Lizard Island. However, present day aggregations reportedly only occur around Townsville, suggesting a southward contraction in the spawning aggregation has occurred. If environmentally driven, this could be consistent with a negative response to temperature.

Southern Oscillation Index

The existence of a correlation between broad-scale climate variables has been documented in many of Australia's commercially important fish and invertebrates and also appears to affect Queensland east coast *S. commerson*. We found that one-year lagged SOI explained approximately 26% of variation in the annual CPUE of *S. commerson* over the 24-year period. Generally speaking, La Nina events that led to higher values of SOI resulted in higher catch rates, while El Nino events resulted in lower catch rates. However, the relationship was not definitive. For instance, during the prolonged La Nina conditions between 1998 and 2001 catch rates were relatively stable and the year of highest SOI (financial year 2010/11) did not result in especially high catches.

Strong and consistently positive values of SOI (La Nina events) are associated with higher rainfall across much of Australia leading to increasing coastal and estuarine productivity that has positive effect on the recruitment of many fish species. Since larval *S. commerson* settle in estuaries after a two to four week pelagic phase, we hypothesised that a similar positive effect on recruitment might be observed in this species. Only weak evidence was found that YCS was affected by SOI (a one-year advanced correlation in Rockhampton). The one year

lagged correlation with CPUE was also inconsistent with hypotheses that SOI affect recruitment or catchability directly. The correlations between SOI and CPUE suggest a general association between the variables where higher values of SOI lead to greater coastal productivity, indirectly benefiting Spanish mackerel.

River flow

Although SOI is often a proxy for increased estuarine productivity, in many species a more direct correlation can be found between recruitment and the volume of water discharged from the river itself (Balston 2009, Halliday *et al.,* 2011). In this study the one-year lagged correlation between CPUE and spring and summer flow of the Herbert river system was significant, however explained only 18% of the variation. Although not presented, a similar statistical correlation was found with the Burdekin River system. These results are concordant with the above findings that suggest La Nina conditions are associated with a general increase in productivity, however they did not provide any more definitive evidence of a mechanism, and the correlation was weaker than with SOI itself.

Chlorophyll-a

A highly significant positive correlation ($r^2 = 0.67$) was found between YCS and spring Chl-a in the coastal grid directly adjacent and inshore of the Townsville spawning reefs. Although also a lagged response, this finding is an interesting outcome and warrants further investigation. Unfortunately, data for this variable was available for only 10 years of the time-series and heavily influenced by the particularly strong year of recruitment in 2009. As such it wasn't possible to draw strong conclusions about the nature of this correlation.

Study limitations

In this study we considered four environment-recruitment hypotheses for *S. commerson.* To help separate causal relationships from general associations between variables, each hypothesis was tested against the year of recruitment itself, one year lagged and one year advanced. A challenge faced was the lack of detailed information on the early life history of *S. commerson* from eastern Australia, making it difficult to develop and test very specific environment-recruitment hypotheses on an appropriate spatial scale. This lack of knowledge was a major factor in the choice to only use single variable linear regressions, as opposed to considering multiple regression models with many variables and interactions. Such models would undoubtedly have led to many more significant correlations, but assessing their biological validity would have been challenging. A better understanding of the dispersal and movement of larval *S. commerson*, and the linkage between the reef and estuary would be useful to help devise more tractable environment-recruitment hypotheses.

Management implications

This study confirms the long-held suspicion that recruitment of *S. commerson* is variable and, at least partly, environmentally driven. In comparison to estuarine species, however, recruitment of *S. commerson* was relatively more stable; the base YCS models with age and sample year as explanatory variables accounted for 71–87% of total variation. This compares to 52–62% for *L. calcarifer* from the Fitzroy river (Staunton-Smith et al. 2004, Halliday et al. 2011) and 55% for king threadfin, *Polydactylus macrochir* (Halliday et al. 2008). Another important outcome of this study was the finding that patterns in recruitment of *S. commerson* were similar over the entire spatial extent of the study area. This supports the current view that fish on the east coast are a single stock for management purposes. This also could suggest that either (a) the majority of fish originate from the main spawning aggregation around Townsville or (b) spawning occurs at various locations but is influenced by the same oceanographic conditions over a broad scale. Given the high degree of collinearity between SST in each region, it wasn't possible to establish which of these situations is most likely at present. The findings of this study also confirm an important link between CPUE and SOI in the Townsville region. Overall we found some degree of support for all of our original hypotheses, and strong support that SST and SOI affect recruitment in particular. Although many of the direct mechanisms by which these variables effect recruitment are not yet clear, they do provide preliminary evidence that management of this species may need to factor in climate variability.

8.5 Vulnerability assessment

8.5.1 Overall vulnerability and potential impacts

The vulnerability assessments for key fishery species of northern Australia are presented below for each of the three regions: north-western Australia, Gulf of Carpentaria and the east coast. Given the framework structure, species with the highest vulnerability will be exposed to changes in the environment, will have a high sensitivity to these changes, and will have a low capacity to adapt. There were several factors that made a species vulnerable to climate change. Generally, species that tended to have the highest exposure were those with an estuarine and/or nearshore distribution (e.g. golden snapper, mud crab, mangrove jack), low mobility (e.g. sandfish, black teatfish, banana prawn), and a dependency during critical parts of their life cycle on habitat types more likely to be impacted by climate change (e.g. tiger prawns, sandfish, tropical rock lobster that are dependent on coral reefs and/or seagrass meadows). Those with low exposure tended to have deep water and/or pelagic habitat preferences (e.g. red emperor, scallops, blacktip sharks, billfish) and high mobility (e.g. blacktip sharks, scalloped hammerhead sharks, billfish, spotted mackerel). This is because nearshore shallow areas are directly exposed to changes in most key climate variables that are predicted to change, including rainfall, riverflow, salinity, SST, pH, and include vulnerable habitats such as seagrass meadows and mangroves. Also, more mobile animals are better able to moderate their exposure to environmental changes, particularly during periods of rapidly changing conditions caused by extreme events such as cyclones and floods.

The environmental variables that north-western Australian and Gulf of Carpentaria fishery species are likely to have greatest exposure to in the future are ocean acidification, reduced salinity, increasing SST and altered rainfall/riverflow. The indirect effects of changes in habitats are also likely to be important. The environmental driver likely to be most influential on the east coast is altered riverflow (rainfall) since there is a known positive correlation between riverflow (rainfall) and abundance/growth/catch rates of several fishery species. Rainfall is projected to decrease on the east coast by 0 to 10% by 2030, however due to the likely increase in water extraction this is likely to be closer to the lower end (-10%) or worse. Increasing SST and habitat changes are also likely to be important drivers that will affect fishery species on the east coast.

Species with a high sensitivity tended to be low productivity species (i.e. late maturing, low fecundity) (e.g. scalloped hammerhead, black teatfish, bull shark) and/or have a known reliance on environmental drivers (e.g. for successful recruitment) (e.g. tropical rock lobster, mud crab, barramundi, king threadfin). Species with low sensitivity to future climate change tended to be highly productive (e.g. blue threadfin, Spanish mackerel, tiger prawns) or deep-water species (e.g. red emperor, goldband snapper). It is worth noting that the

sensitivity of individual species to particular environmental variables is poorly understood overall. However, where knowledge was relatively good for a particular species they tended to receive a higher sensitivity score, while other species where information was lacking may have received a lower sensitivity score. Expert opinion was incorporated into the assessment process in an attempt to address this potential bias.

Species with a low adaptive capacity tended to be those that are overfished to some degree (e.g. golden snapper, king threadfin, black teatfish, black jewfish), have low productivity (e.g. golden snapper, mangrove jack, bull shark, scalloped hammerhead shark) and have low mobility (e.g. Moreton Bay bug, black teatfish, sandfish). Species with high adaptive capacity tended to be those with high mobility. The full scores for Exposure, Sensitivity and Adaptive Capacity indicators, as well as the direction of impact, for each of the three regions are in Appendix 7.

8.5.2 Individual species vulnerability for 2030

Based on the species reviews and expert opinion, the likely impacts of climate change on key northern Australian fishery species by 2030 are provided in Table 8.33. This table was central in informing the vulnerability assessment and also the scenarios provided to stakeholders for identifying adaptation options that were relevant (see Section 8.6).

There were 23 species included in the ecological vulnerability assessment for north-western Australia based on projected climate change for 2030. The three species with the highest vulnerability were golden snapper, king threadfin and sandfish while mangrove jack also had a high relative vulnerability score (Figure 8.40). The least vulnerable species were the shark species (with the exception of bull shark and pigeye shark) and the pelagic Spanish mackerel and sailfish. Golden snapper are one of the most important recreational species in the Northern Territory where they are at the northern limit of their range in Australia. King threadfin are also relatively important both commercially and recreationally.

In the Gulf of Carpentaria there were 21 species included in the vulnerability assessment for 2030. The species with the highest relative vulnerability were golden snapper, king threadfin, sandfish, tiger prawn, mangrove jack and banana prawn. Similar to north-western Australia the least vulnerable species were sailfish and Spanish mackerel, as well as the blacktip shark (*C. limbatus* only), spot tail shark and scalloped hammerhead shark (Figure 8.41).

On the east coast there were 24 species included in the final vulnerability assessment. Two species had much higher vulnerability scores than any other: black teatfish and king threadfin. Other species that had moderately high vulnerability were sandfish, barramundi, tiger prawn, golden snapper, white teatfish, banana prawn and mangrove jack. The least

vulnerable were Moreton Bay bug, spotted mackerel, black marlin and eastern king prawn
(Figure 8.42).

Table 8.33 Likely impacts on key northern Australian fishery species based on climate change projections for 2030 (A1B & A1FI).

Species	Key potential effects of climate change (based on 2030 projections)
Banana prawn	• Sea-level rise may increase/decrease abundance due to alteration of mangrove habitat availability, depending on local barriers for mangrove replenishment and migration (e.g. coastal development) (+/-) • Altered rainfall will likely result in concomitant changes in population abundance (slight increase in NWA and GoC, decrease on the EC)(+/-) • Increasing SST will likely result in a poleward distributional shift into NSW waters on the EC (+/-)
Eastern king prawn	• Changes in the EAC and onshore wind patterns may affect larval movement and recruitment (+/-) • Increased SST may result in lower abundance in the SE Queensland region (-) • Increasing SST may result in a poleward range contraction from SE Queensland (-ve for Qld, +ve for NSW)
Tiger prawns (Brown and Grooved)	• Predicted negative impacts on seagrass beds may reduce abundance due to decreased juvenile growth and survival (-) • Increasing SST may compromise growth and survival of Torres Strait stock of brown tiger prawn as they are near their northern range limit (-) • Altered rainfall may affect the catchability of tiger prawns (slight increase in NWA and GoC, decrease on the EC) (+/-)
Mud crab	• Increased SST may result in higher catch rates (+) • Altered rainfall (riverflow) and may increase mud crab abundance in NWA and GoC, and decrease abundance on the EC (+/-) • Sea-level rise may increase/decrease abundance due to alteration of mangrove habitat availability, depending on local barriers for mangrove replenishment (eg. coastal development) (+/-)
Sandfish	• The effects of climate variables on sandfish life history stages are poorly understood. • Predicted impacts on seagrass meadows may affect survival of juvenile sandfish as it is their preferred habitat for settlement.
Saucer scallop	• Increasing SST may result in poleward movement of SE Qld spawning grounds or into deeper water as spawning occurs at the coolest time of year (-) • Lower rainfall in SE Qld may reduce recruitment (-) • Changes in major currents (Leeuwin in WA, EAC on EC) may impact recruitment success (+/-)
Black teatfish	• Increasing SST may compromise reproductive success since they spawn during winter in far northern areas, e.g. Torres Strait, resulting in range contraction poleward (-)

Species	Key potential effects of climate change (based on 2030 projections)
Tropical rock lobster	• Increasing SST may promote faster growth and higher larval supply, but may decrease juvenile survival. The net result may be a reduction in spawning biomass (-) • Adults are likely to move to deeper to less accessible fishing areas or father south with increases in water temperature (and extremes) (-) • Changes in currents in the northwest Coral Sea may alter settlement areas and recruitment rates (+/-)
Barramundi	• Altered rainfall may affect the abundance, growth and catchability of barramundi (slight increase in NWA and GoC, decrease on the EC). A potential increase in the GoC may be offset by proposed water extraction from Gulf rivers for land use (+/-) • Sea-level rise may alter the availability of suitable floodplain nursery areas for post-larvae and juveniles: NWA and GoC (+/-) and EC (-) • Increased variation in rainfall may reduce the frequency of large flood events reducing overall population sizes on the EC. Longer periods of drought predicted for the east coast could significantly reduce barramundi populations (especially periods > ~7 years). This is likely to be exacerbated by increased water extraction for land use (-)
Coral trout – common/barcheek/ passionfruit	• Increases in intense storm activity may periodically reduce the catchability of coral trout (-) • Increased water temperatures (particularly in areas where SST exceeds 30 °C) may reduce survival and development of egg and larval stages resulting in lower population sizes. Adults may also move poleward or to deeper water: In northern regions (-); in southern regions (+) • Increased SST compromising coral reef habitat may affect juvenile survival (-) • Spawning may occur earlier than currently (region-specific) (+/-)
Golden snapper	• Potential range expansion poleward on the east and west coasts (+) • Relationships with climate variables poorly understood however their resilience to future changes may be poor due to their late maturity and overfished status in some areas (-)
King threadfin	• Altered rainfall may affect the recruitment and abundance of king threadfin (slight increase in NWA and GoC, decrease on the EC) (+/-) • Increased SST may result in a range extension poleward on the east and west coasts (+) • Resilience to future changes may be poor due to their large size and older age at sex change (to female) and overfished status in some areas (-) • Localised population impacts may be evident due to their fine scale stock structure (+/-)

Species	Key potential effects of climate change (based on 2030 projections)
Red throat emperor	• Increases in intense storm activity may periodically increase the catchability of red throat emperor (+) • Increasing SST may result in a range shift poleward associated with a contraction of the northern range limit (+/-)
Spanish mackerel	• Increasing strength of the EAC likely to cause a poleward range extension (+ve for SE Qld/NSW & SW WA) • Increasing SST could also cause a poleward shift of the main spawning (and fishery) area on the east coast and/or lower east coast population sizes (+/-)
Mangrove jack	• Altered rainfall may affect the juvenile survival and therefore population abundance; NWA and GoC (+/-) and EC (-) • Sea-level rise may alter the availability of suitable floodplain nursery areas for juveniles: NWA and GoC (+/-) and EC (-)
Black jewfish	• Altered rainfall may affect the juvenile survival and therefore population abundance; NWA and GoC (+/-) and EC (-) however this is poorly understood for this species • Their current overfished status in all regions reduces their resilience to cope with potential negative impacts of climate change (-)
Grey mackerel	• Altered rainfall may affect the juvenile survival and therefore population abundance; NWA and GoC (+/-) and EC (-) however this is poorly understood for this species
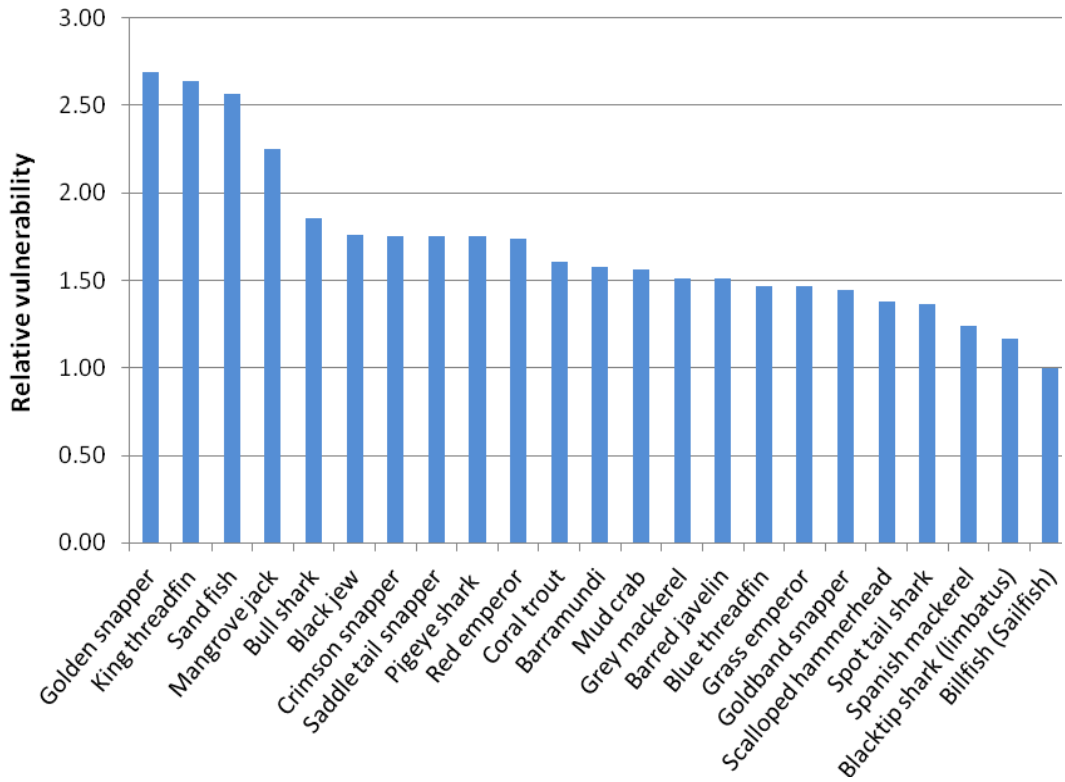
Figure 8.40 Relative vulnerability scores for key fishery species of north-western Australia (2030).
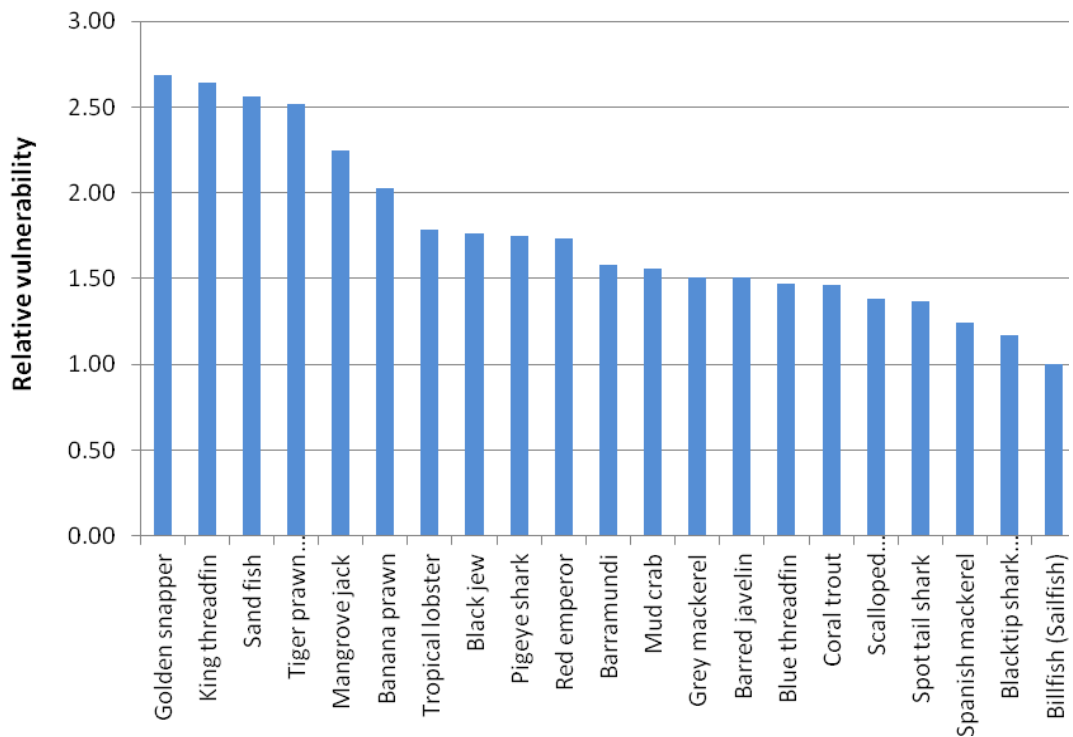
Figure 8.41 Relative vulnerability scores for key fishery species of the Gulf of Carpentaria (2030).
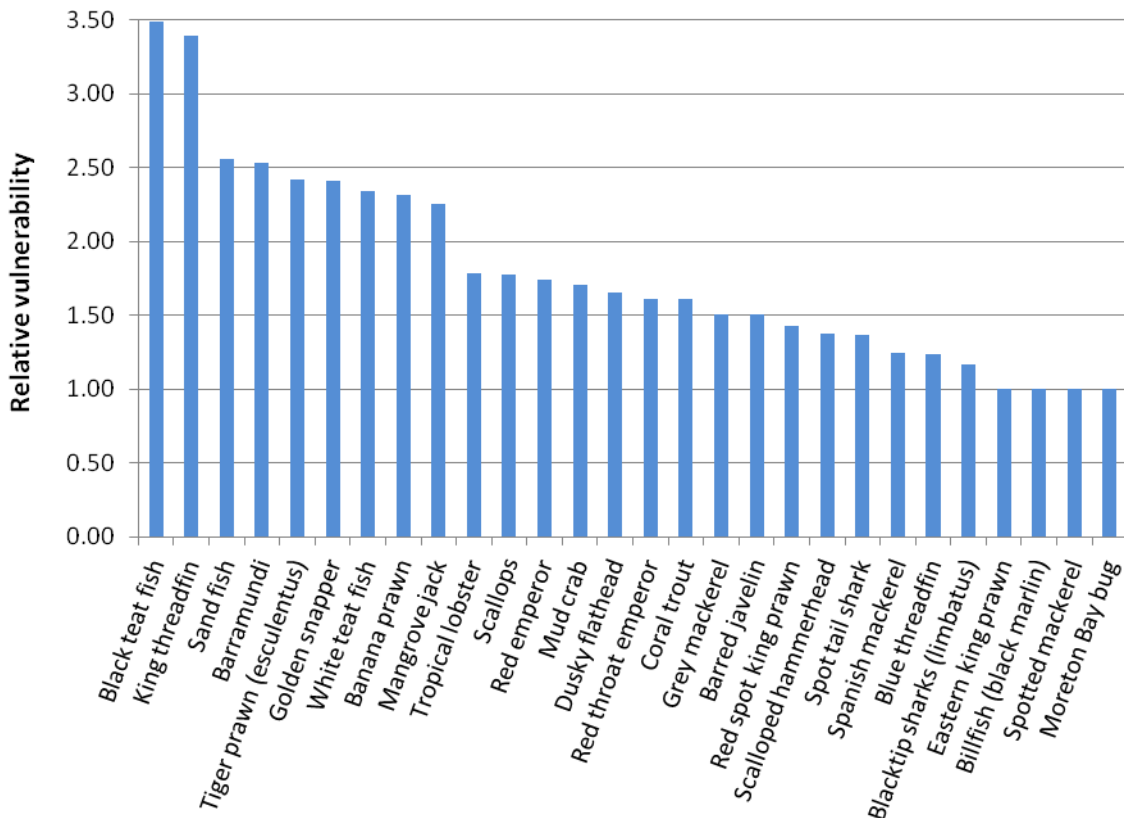
Figure 8.42 Relative vulnerability scores for key fishery species of the tropical east coast (2030).

8.5.2.1 King threadfin

High vulnerability. Similar to golden snapper, king threadfin had very high vulnerability scores in all regions. They have high exposure and low adaptive capacity, but with a relatively higher sensitivity primarily due to their known reliance on riverflow as a driver of recruitment. Compared to golden snapper they have greater replenishment potential overall, notwithstanding their large size and late age at sex change to female, however they are also potentially overfished in some areas suggesting that prudent fisheries management intervention is needed including setting appropriate size limits.

By 2030 the projected increase in rainfall (and riverflow) in north-western Australia and the Gulf of Carpentaria may benefit king threadfin populations. However, the projected increase is small (0 to +5%) and any increase may be offset by future water extraction for land-based use. On the east coast, rainfall (and riverflow) is projected to show a decrease by 2030 (0 to -10%) with water extraction also likely to increase from current levels, possibly resulting in a decrease in king threadfin populations. Given projected increases in SST there is the potential for poleward range extensions on the east and west coasts. Given the recent example in the Brisbane River where king threadfin numbers have increased in recent years and may be linked to improving water quality, any southerly range extension may depend on local estuary water quality characteristics. Monitoring projects such as Redmap

(www.redmap.org.au) may help to shed more light on future range extensions of this and other tropical species.

8.5.2.2 Black teatfish

High vulnerability. Black teatfish were only assessed on the east coast where they have historically been the main target species for the east coast sea cucumber fishery on the Great Barrier Reef. However, fishing has been banned since 1999 due to overfishing. Despite this history, their continued high value and demand from SE Asian countries justified their inclusion for the east coast assessment.

Black teatfish had the highest relative vulnerability score for the east coast. Although they are a reef-based species generally found offshore, they are highly exposed to increasing SST and intense cyclones due to their shallow water preferences and low mobility, resulting in moderate exposure. Their overall high vulnerability comes from having high sensitivity to future projected climate change and low adaptive capacity. Black teatfish are strongly associated with coral reef habitats, which are projected to be negatively impacted by 2030, and they spawn in winter so increasing SST may cause animals to cease spawning in northern areas (e.g. Torres Strait/far northern GBR) with a possible poleward range contraction. They have low adaptive capacity due to their low mobility, their status as overfished in the GBR, and an apparent low replenishment potential.

The key fisheries for black teatfish in northern Australia are currently closed due to overfishing, and have been for some time. Given the lack of recovery evident and the outlook based on this assessment, the future of fisheries for black teatfish in Australia is not promising.

8.5.2.3 Golden snapper

High – moderately high vulnerability. Golden snapper had the highest, or near highest, relative vulnerability scores for all three regions. Across northern Australia, and particularly around Darwin in north-western Australia, they represent an iconic and highly sought after target species. They have high vulnerability to projected climate change because they will be highly exposed due to their nearshore habitat preference, and particularly the preference for juveniles (and possibly larvae) to occupy estuaries. They also have very low adaptive capacity because they are experiencing localised overfishing in many areas, have low replenishment capacity (late maturing), and may be subject to non-fishing pressures given their estuarine/nearshore habitat preference.

No studies have explored the importance of environmental variability on the different life history stages of golden snapper, somewhat hindering the ability to confidently score assessment indicators. Given projected increases in SST there is the potential for poleward

range extensions on the east and west coasts, although the availability of preferred habitats may be a limiting factor. Given their overfished status in some areas prudent fisheries management intervention is needed especially the implementation of appropriate size limits. Maintenance of healthy nearshore/estuarine habitats is also likely to be critical for golden snapper and many other key fishery species.

8.5.2.4 Sandfish

High – moderately high vulnerability. Sandfish also had a high vulnerability in all three regions. Their level of harvest has historically been variable across northern Australia however they are targeted in all regions, being the predominant species (historically) in the Northern Territory and Western Australia. They are highly exposed to environmental changes because they have a nearshore/shallow water habitat preference. They have moderate sensitivity to environmental changes and very low adaptive capacity. Sandfish juveniles rely on seagrass meadows for settlement and feeding, which has implications for the species since seagrass habitats are predicted to decrease in area by 2030. This is likely to have a knock-on effect reducing sandfish populations. Their low adaptive capacity is influenced by their low mobility but also by the fact that they are locally overfished in some areas (e.g. Torres Strait) and have a low replenishment potential.

Generally, very little is known about the effects of climate variability on sandfish populations. Given their shallow nearshore habitat preferences, increases in SST, ocean acidification and reduced salinity may negatively impact on larval development of sandfish (and other sea cucumbers) however the time horizon for such effects are uncertain. Management options are for the conservation and rehabilitation of seagrass habitat.

8.5.2.5 Tiger prawn

High – moderately high vulnerability. Tiger prawns are comprised of two species: the brown tiger prawn and the grooved tiger prawn. The brown tiger prawn is the dominant species and so was the focus of this assessment. Tiger prawns were not assessed in north-western Australia where fisheries don't operate. They represent a key species in the Gulf of Carpentaria Northern Prawn fishery (along with banana prawns) and in the east coast trawl fishery. Tiger prawns were assessed as highly vulnerable to climate change by 2030 in the GoC and moderately highly vulnerable on the east coast. This is mainly due to the fact that they have very high exposure to almost all environmental variables predicted to change, as well as changes in seagrass habitats, which provide critical habitat during the juvenile life history stage. Tiger prawns are only moderately sensitive to environmental change mainly due to their high productivity and rapid rate of population turnover, however their reliance on seagrass meadows as juveniles means that likely impacts on seagrass will reduce juvenile growth and survival and ultimately reduce tiger prawn population sizes. Obvious

management options involve the conservation and rehabilitation of seagrass habitats, particularly as they serve as important nursery areas for many other fishery species.

8.5.2.6 Mangrove jack

Moderately high vulnerability. Mangrove jack were assessed as having moderately high vulnerability to climate change, due to their moderately high exposure and sensitivity, and moderately low adaptive capacity. They will be highly exposed due to their nearshore and estuarine habitat preference during the critical juvenile phase. They have a similar life history to golden snapper and are therefore sensitive to environmental changes for similar reasons, although their reliance on environmental drivers for spawning/settlement increases their sensitivity. They have higher adaptive capacity than golden snapper as they are not considered overfished and the availability of suitable habitat outside their current range is very high. The likely impacts of climate change on mangrove jack are unknown due to a lack of relevant research.

8.5.2.7 Barramundi

Moderate – High vulnerability. Barramundi were assessed as moderately vulnerable in north-western Australia and the Gulf of Carpentaria, and as highly vulnerable to climate change on the east coast. Overall, barramundi will be highly exposed due to their nearshore and estuarine habitat preference, particularly given their close association with river systems at all life history stages. They tend to be adaptable to changes and have been shown to have a wide thermal tolerance (Jerry et al. 2014), however have a known strong reliance on rainfall (and associated linkages: riverflow, flood plain inundation, higher nutrient levels/food availability) as a trigger for spawning but also for downstream movement of adults and upstream movement of juveniles into floodplain areas. Riverflow also influences growth rates and fishery catch rates all of which may be enhanced in north-western Australia and the GoC by projected slight increases in rainfall. Further, increased rainfall and rising sea level may increase the availability of important flood plain habitat that would enhance juvenile survival. However, this will be dependent on local factors such as coastal development and topography, which may prevent landward migration of habitat as sea level rises, and the extent of water extraction for land use.

Although their ecological adaptive capacity is generally high, they have a high dependence on riverflow (rainfall) for growth, and for facilitating juvenile survival as recruits to the next generation. Riverflow also enhances fishery catch rates. Barramundi vulnerability in the three regions across northern Australia is largely influenced by regional projections in rainfall (riverflow). Rainfall is projected to decrease on the east coast and water extraction is projected to also increase, exacerbating a future of lower riverflow. Further, coastal development on the east coast is far more progressed than in other parts of northern Australia and is likely to limit any positive effects of sea-level rise on flood plain inundation.

On the east coast, localised impacts on barramundi populations into the future could be significant.

8.5.2.8 White teatfish

Moderately high vulnerability. White teatfish have a similar life history to black teatfish, however they were assessed as being less vulnerable to climate change due to several factors. Firstly, white teatfish are likely to be less exposed due to a preference for deeper reef slopes. White teafish are not a winter spawner and so are less likely to be sensitive to increasing SST. Their adaptive capacity is likely to be greater, since they are not overfished and remain a target species in the Great Barrier Reef sea cucumber fishery. Despite this, the effects of climate change on white teatfish are poorly understood and their close association with coral reefs and low mobility contribute to their assessment as moderately high vulnerability.

8.5.2.9 Banana prawn

Moderately high vulnerability. Possible opportunity species. Banana prawns were assessed in the Gulf of Carpentaria and on the east coast where they have a moderately high vulnerability. Like many other prawn species, banana prawns have a high exposure to most of the key environmental variables that are projected to change. Their sensitivity is higher than other prawns however largely due to their known reliance on rainfall for recruitment. In years of high rainfall (and high riverflow) banana prawn recruitment and catches are greater. Further, banana prawns use mangroves as important juvenile habitat. Projected sea-level rise impacts on mangroves vary by region, depending on local topography and coastal development. Although mangrove habitat areas may shift inland from current locations, it is unlikely that the total habitat area will increase and in some areas may decrease depending on barriers to migration. It is possible that there will be an increase in banana prawn population sizes by 2030 in the Gulf of Carpentaria due to increased mangrove habitat, and predicted slight increases in rainfall that will not only enhance recruitment but also catchability. On the east coast however, there is more likely to be a decrease in banana prawn populations by 2030 due to lower rainfall and the higher likelihood that mangrove habitat area will decrease due to sea-level rise and coastal development.

8.5.2.10 Mud crab

Moderate vulnerability. Possible opportunity species. Mud crabs are widespread across tropical and sub-tropical Australia and are popular target species throughout this range. They were assessed as being moderately vulnerable to climate change by 2030 in all three regions. Mud crabs have high exposure mainly due to their preference for shallow nearshore/estuarine environments. They also have high sensitivity meaning the potential

impacts of climate change on populations could be high. Although they are not very mobile they can be highly productive and have a very wide range across northern Australia giving them moderately high adaptive capacity.

By 2030 mud crab populations in north-western Australia and the Gulf of Carpentaria are likely to benefit from increased rainfall (and riverflow), and increased SST that is likely to enhance growth rates and fishery catch rates. Mud crabs have been shown to use both seagrass meadows (crablet stage) and mangrove forests (juvenile stage) during their early life history. Increases in sea level will probably increase the availability of mangrove habitat in the GoC, potentially resulting in larger mud crab population sizes. This may be negated in areas where local coastal development restricts mangrove migration. On the east coast, increasing SST is also likely to enhance mud crab growth and fishery catch rates. However, increases in sea level will potentially decrease the availability of mangrove habitat in some areas, and lower rainfall (riverflow), exacerbated by greater water extraction for land use, will likely offset any positive effects on populations and may result in lower population sizes by 2030.

The management implications for this in northern Australia are that fishery catches may be greater in the Gulf of Carpentaria and north-western Australia, while on the east coast it may remain similar or lower depending on local rainfall patterns. One of the key unknowns is the effect that ocean acidification will have on mud crabs. This has the potential to significantly impact on crab development and survival at all life history stages. This makes mud crab one of the high priority species for investigating the effects of lowered ocean pH.

8.5.2.11 Coral trout

Moderate vulnerability. Coral trout were assessed as being moderately vulnerable to climate change by 2030. This is mainly because, as a primarily offshore and sometimes deep-water species, they are less exposed to environmental change compared to shallow nearshore species. They are also a highly productive species, relatively mobile and have a wide distribution. Despite these attributes and this assessment, the vulnerability of coral trout to climate change beyond 2030 is likely to be greater. Recent research has highlighted that SST above 28°C negatively impacts on the development of early life history stages (Pratchett et al 2013). Coral trout may be able to adapt to this by spawning earlier than they do currently, however northern-most populations are likely to be affected in the medium-term as this critical temperature threshold is reached (see Pratchett et al 2013). Coral trout also use coral reefs as key habitats throughout their life history stages. Coral reef habitats are predicted to be impacted by 2030, which is likely to have indirect impacts on coral trout populations due to the reduced availability of preferred juvenile habitat that is likely to reduce juvenile survival (Pratchett et al 2013). Increased intensity of cyclones is likely to result in more frequent periods where catchability is reduced (Tobin et al 2010). Although

from an ecological perspective this is not likely to be a threat to coral trout, from a fishery perspective this would have significant economic and social impacts due to the high reliance on operations specialised for live product. Increasing acidification will also impact coral trout populations through lower survival of juveniles (Munday et al 2012), however the pH levels at which this is likely to occur will not be evident until later this century. One of the key research questions this raises is the capacity of coral trout to adapt to altered conditions (e.g. elevated SST) in the future. The interesting and surprising result of acidification on coral trout early development and survival also raises questions of how other species may be affected by pH. Similar research to that by Pratchett et al (2013) on acidification should be carried out on other key fishery species.

Given the potential impacts identified for coral trout, and the highly targeted nature of the fishery, future fishery operations may need to be diversified in terms of target species. Coral trout are likely to remain in demand throughout south-east Asia, however in the event of lower supply potential other less valuable species will need to be considered. The profitability of fishing businesses would also suffer unless operational costs can be reduced using, for example, cheaper fuel options. Given the importance of this fishery to all sectors, on the Great Barrier Reef in particular, fishery-specific in depth analysis of adaptation options with stakeholders is warranted.

8.5.2.12 Tropical rock lobster

Moderate vulnerability. Tropical rock lobster (TRL) was assessed in the Gulf of Carpentaria and the east coast and was moderately vulnerable. They are highly exposed to climate change by 2030 mainly because they use both coral reefs and also nearshore habitats, and have relatively low mobility. Although their overall sensitivity to environmental change is low, they are highly sensitive to increases in SST. In far northern areas of their range (e.g. Torres Strait) TRL already show a response to warm ocean conditions by moving into deeper water, with negative consequences for fisheries. They may also begin to range shift further south. From a fishery perspective, there are no comparable alternative species given the market price and demand for rock lobster and the high degree of gear and fishing technique specialization. Managing the fishery to mitigate these types of potential impacts is difficult, however fishery participants in the Torres Strait in particular should be consulted regarding potential implications of climate change and future actions.

One of the key unknowns that may impact on TRL populations, is the effect climate change will have on ocean circulation in the western Coral Sea (Coral Sea gyre). Given the role this current plays in the transport of larvae and the duration of oceanic larval development, recruitment of TRL to Australian waters could be impacted significantly. Improved understanding of the likely future ocean circulation in the north-western Coral Sea is therefore important future research needed.

8.5.2.13 Saucer scallops

Moderate vulnerability. Scallops were assessed on the east coast only since this region has an important fishery operating in a narrow area of south-eastern Queensland with saucer scallop the major target species. They are found in relatively deep water in offshore sandy habitats and for this reason have a low exposure to environmental changes. They are moderately sensitive to environmental changes, which is influenced by their likely reliance on certain environmental conditions for successful recruitment. This appears to be quite complex, as evidenced by the exhaustive analyses on this species carried out during this project, with SST, local currents and local riverflow (rainfall) potentially important. Cooler SST appears to favour high recruitment and so increasing SST over time may have the effect of reducing the population and/or causing a poleward range shift into NSW waters and/or populations moving deeper in south-east Queensland. Riverflow also appears to have a positive effect on recruitment, probably through increased nutrient loads and therefore food availability at the critical early life history stages. Projections of lower rainfall on the east coast may therefore be detrimental to scallop populations. The annual presence of a cyclonic current eddy in the Capricorn region was also shown to positively influence scallop recruitment. It was not clear from the analyses however, how important the timing of this eddy is, and the effects of climate change on local current patterns are poorly understood. A better understanding of this through hydrodynamic modelling would be useful research as any changes in local current patterns could significantly affect scallop recruitment. In the meantime, continued monitoring of annual recruitment and catch in the fishery will help to detect any changes in the future such as those postulated here.

8.5.2.14 Bull shark/Pigeye shark

Moderate vulnerability. Bull shark were not initially identified as being a key fishery species in northern Australia, however since the project inception it has been discovered that at least half (approximately) of the NT pigeye catch is actually bull shark. Bull shark was therefore included in the vulnerability assessment for north-western Australia. Of all the shark species assessed bull shark and pigeye sharks had the highest vulnerability with the rest having low vulnerability. Although moderately exposed, largely due to their close association with estuarine and/or nearshore habitats, and with a high adaptive capacity, both species have a high sensitivity to climate change. This is largely due to their very low productivity, typical of many elasmobranch species, but also because they use nearshore and estuarine waters as important pupping habitat, probably as feeding and predator avoidance strategies. Bull shark in particular use estuaries and upper reaches of rivers as juvenile areas. Continued monitoring of the relative catch of each species is therefore warranted as well as management consistent with low productivity species.

8.5.2.15 Spanish mackerel

Low vulnerability. Possible opportunity species. Spanish mackerel are important in all regions of northern Australia and were consistently assessed as having low vulnerability to future climate change. They have a moderately high exposure, mainly due to their use of nearshore habitats during much of their life, and the use of estuaries by juveniles. To offset this exposure they have moderately low sensitivity and high adaptive capacity. Spanish mackerel are highly productive and mobile with a range of habitat preferences and bait types making them less vulnerable overall relative to other species. The key important drivers are likely to be increased SST and possibly changing currents and rainfall. Higher SST may result in smaller population sizes of Spanish mackerel in eastern Australia, since the preliminary analyses conducted during this project suggest higher recruitment at lower spawning temperatures. It may also result in a later onset of spawning or a poleward shift of the east coast spawning (and fishery) grounds. Similar impacts are possible on the west coast are likely as SST increases. This could also mean lower population sizes in the future for Gulf of Carpentaria and Northern Territory fisheries.

The effect of rainfall and riverflow on recruitment was also investigated but produced equivocal results, however given juveniles spend several months in estuaries rainfall is likely to have some influence on recruitment. Increasing SST, along with the projected strengthening of the East Australian Current, will likely result in a poleward range extension. This would result in longer fishing seasons and higher catches in south-east Queensland and NSW, providing a potential opportunity for these fisheries. This presents management implications that involve better inter-jurisdictional co-operation and should at least involve close monitoring of catch levels in these regions over the coming years. It may also require a review of current management arrangements in NSW. Future research should assess the relationship between the strength of the East Australian Current and catches in SE Queensland and NSW.

8.5.2.16 Grey mackerel

Moderate vulnerability. Grey mackerel are a very important fishery species in all three regions and were assessed as being only moderately vulnerable to climate change by 2030. They have a moderately high exposure largely due to their nearshore habitat preferences and because juveniles prefer estuarine systems. Unusually, their distribution at certain times of the year is unknown, on the east coast in particular, making the assessment of exposure slightly uncertain. Their sensitivity is low, largely because it is poorly understood and it is highly possible that populations are influenced by rainfall (riverflow) events given larvae develop in nearshore areas and juveniles occupy estuaries during the annual wet season. For the east coast, where they have been community concerns in recent times about their sustainability, this may result in lower population sizes of grey mackerel by 2030 and beyond. Although commercial catch limits were introduced in Queensland recently,

these results suggest careful monitoring of population levels into the future. ON a positive note, they are a species with high adaptive capacity due to their high productivity and high mobility. Although it is a reasonable assumption that grey mackerel recruitment is influenced by riverflow, as has been demonstrated for several other species with similar habitat use during their life histories, future research efforts should attempt to demonstrate this relationship.

8.5.2.17 Black jewfish

Moderate vulnerability. Black jewfish were assessed in north-western Australia and the Gulf of Carpentaria, where they are targeted by fisheries, primarily recreational. They have moderately high exposure because they occupy nearshore and estuarine habitats throughout their life. Their sensitivity was assessed as moderate however this is largely due to the fact that there is a very poor understanding of the influence of environmental variability on black jewfish population dynamics. Given that they occupy nearshore areas and juveniles live in estuaries, rainfall and riverflow are likely to be important drivers. This could mean a positive impact of climate change in the regions assessed. Black jewfish also have moderate adaptive capacity, despite being assessed as overfished across all areas of northern Australia, suggesting prudent fisheries management intervention is needed. Future research should focus on the importance of environmental variability on the life history stages of black jewfish.

8.5.3 Prioritising species for action

Due to the greater uncertainty in climate projections beyond 2030 (largely due to the unknown global response to reduce carbon emissions), and also in the responses of fishery species to the changes that do occur, we focused on the 2030 vulnerability assessment results for prioritising species for action. This near-term time frame is also more meaningful for all stakeholders in terms of making decisions based on these results. Actions in response to the below prioritisation will need to depend on the species and their particular attributes that make them vulnerable. For example, species that were considered to be overfished in these assessments (based on previous assessments and stakeholder views) would be obvious candidates for fisheries managers to review current harvest rates. Actions taken will vary and could include increased and/or introduction of fishery monitoring, revision and/or adjustment of current fisheries management, or changes in fishery targeting and/or operations.

Prioritisation of species from north-western Australia highlighted four species as highest priority: king threadfin, golden snapper, sandfish, and mangrove jack (Figure 8.43). King threadfin and golden snapper are currently key fishery target species in north-western Australia, while sandfish and mangrove jack are less targeted. King threadfin and golden snapper therefore stand out as key species for action in this region.
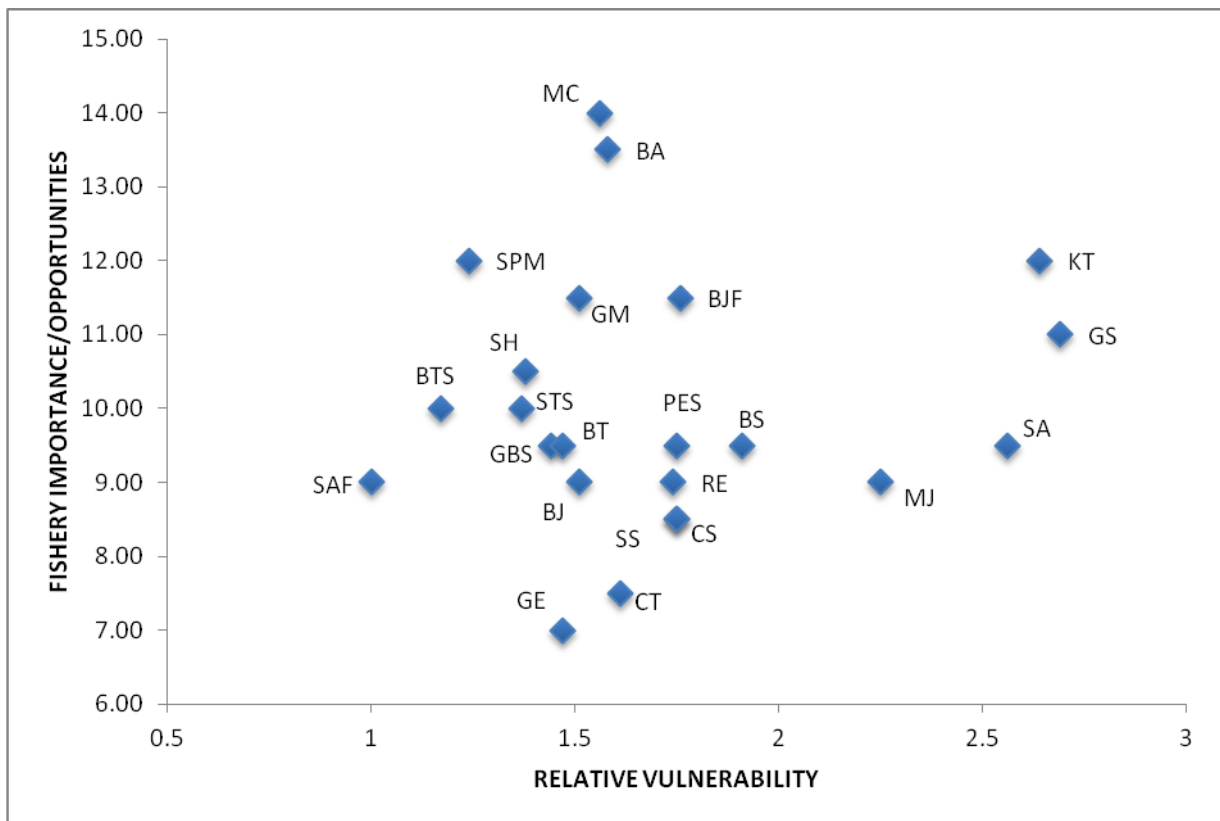
Figure 8.43 Relative vulnerability plotted against the level of fishery importance to assist managers and other fishery end-users in prioritising species for future action – north-western Australian species. **High vulnerability and high fishery importance species are the highest priority (top right of the graph). Species codes are: GS – golden snapper, KT – king threadfin, BJF – black jewfish, SA – sandfish, MJ – mangrove jack, CT – coral trout, GM – grey mackerel, SPM – Spanish mackerel, MC – mud crab, BA – barramundi, RE – red emperor, BT – blue threadfin, BJ – barred javelin, BS – bull shark, PES – pigeye shark, STS – spot tail shark, GE – grass emperor, CS – crimson snapper, SS – saddle tail snapper, SAF – sailfish, GBS – goldband snapper, SH – scalloped hammerhead shark, BTS – blacktip shark.**

Prioritisation of species from the Gulf of Carpentaria also highlighted four species as highest priority: king threadfin, tiger prawn, golden snapper, and banana prawn (Figure 8.44). King threadfin are a key target species for both commercial and recreational fishery sectors as well as having high vulnerability to climate change by 2030. While tiger prawns and banana prawns are the mainstay species of the economically important Northern Prawn fishery as well as having high-moderately high vulnerability. King threadfin, tiger prawns and banana prawns therefore stand out as key species for action in this region. Golden snapper, although having high vulnerability, are less targeted in the GoC and therefore considered of relatively lower priority, although it should be highlighted that they were assessed as overfished for this region.
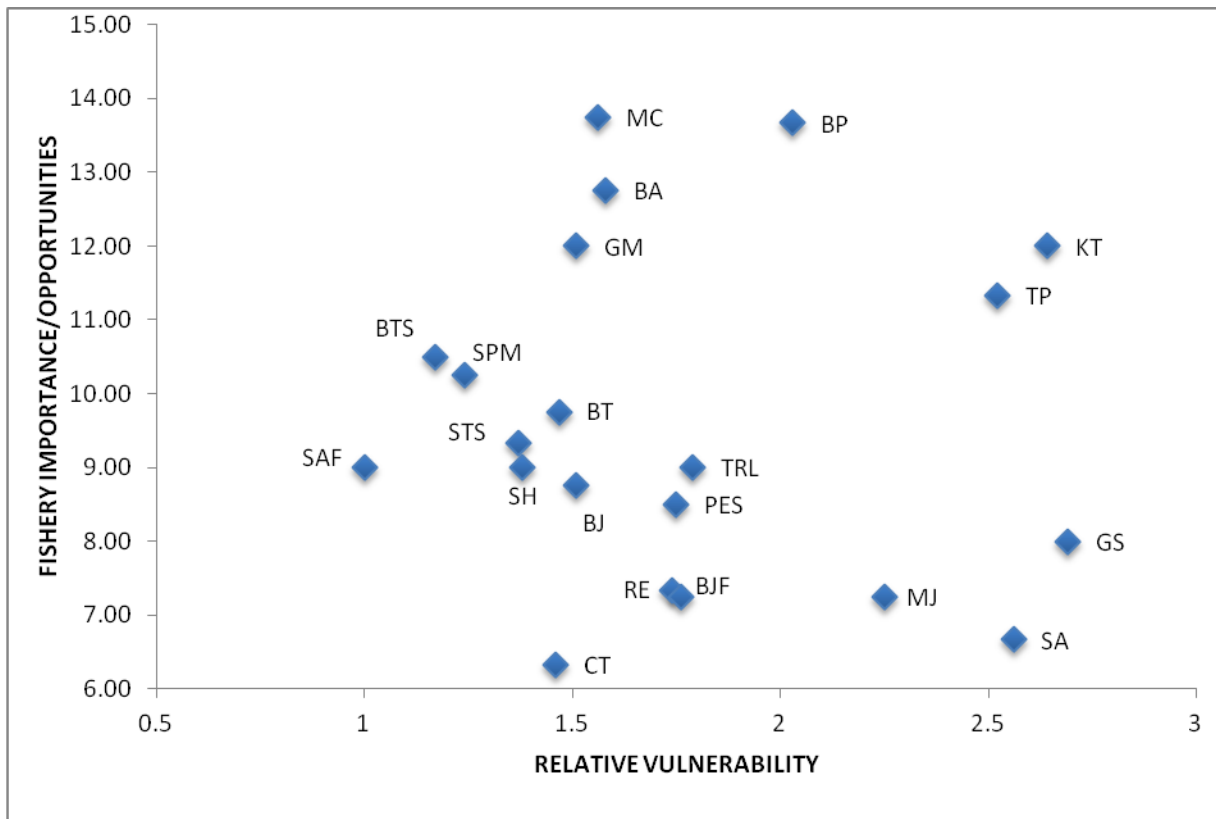
Figure 8.44 Relative vulnerability plotted against the level of fishery importance to assist managers and other fishery end-users in prioritising species for future action – Gulf of Carpentaria species**. High vulnerability and high fishery importance species are the highest priority (top right of the graph). Species codes are: GS – golden snapper, KT – king threadfin, BJF – black jewfish, SA – sandfish, MJ – mangrove jack, CT – coral trout, GM – grey mackerel, BP – banana prawn, TP – tiger prawn, SPM – Spanish mackerel, MC – mud crab, BA – barramundi, RE – red emperor, BT – blue threadfin, BJ – barred javelin, PES – pigeye shark, STS – spot tail shark, SAF – sailfish, SH – scalloped hammerhead shark, BTS – blacktip shark, TRL – tropical rock lobster.**

Prioritisation of species from the east coast highlighted five species as highest priority: king threadfin, black teatfish, barramundi, tiger prawn, and banana prawn (Figure 8.45). All five of these species are important fishery species, however, the fishery for black teatfish has been closed since 1999 and is still closed due to their overfished status. There is still a high demand for this high value species and therefore this assessment process, if nothing else, highlights the importance of maintaining the current fishery closure and the need to monitor for evidence of any recovery. Barramundi and the two prawn species represent important fishery species both economically, and also socially in the case of barramundi. These three species therefore stand out as key species for action in this region. King threadfin are more important at local levels on the east coast, particularly in the Fitzroy River region near Rockhampton, and therefore are relatively lower priority.
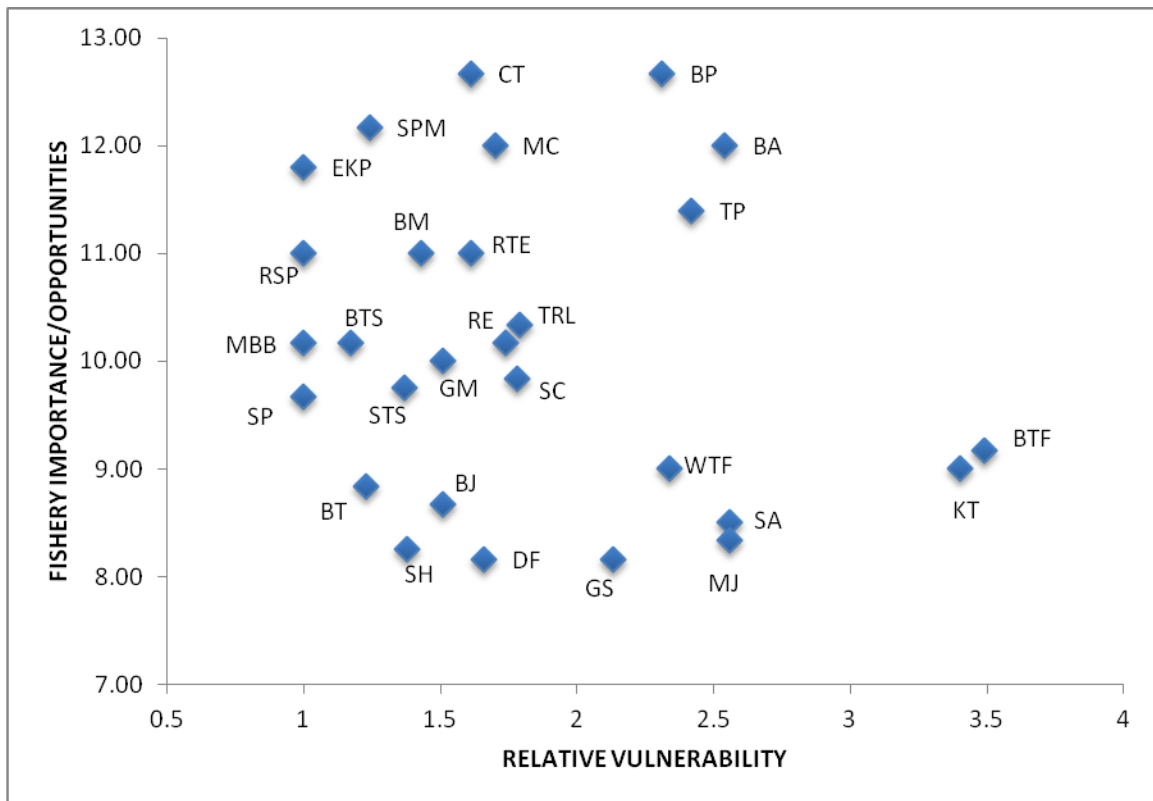
Figure 8.45 Relative vulnerability plotted against the level of fishery importance to assist managers and other fishery end-users in prioritising species for future action – east coast species. **High vulnerability and high fishery importance species are the highest priority (top right of the graph). Species codes are: GS – golden snapper, KT – king threadfin, SA – sandfish, MJ – mangrove jack, CT – coral trout , GM – grey mackerel, BP – banana prawn, TP – tiger prawn, SPM – Spanish mackerel, MC – mud crab, BA – barramundi, RE – red emperor, BT – blue threadfin, BJ – barred javelin, STS – spot tail shark, SH – scalloped hammerhead shark, BTS – blacktip shark, TRL – tropical rock lobster, DF – dusky flathead, WTF – white teatfish, BTF – black teatfish, SP – spotted mackerel, SC – saucer scallop, RTE – red throat emperor, EKP – eastern king prawn, BM – black marlin, RSP – red spot king prawn, MBB – Moreton Bay bug.**

8.5.4   Vulnerability assessments for 2070

The vulnerability assessments for 2070 were based on one future SRES emissions scenario: A1FI. This is the highest future emissions scenario and was chosen for two reasons. Firstly, the most recent assessment (IPCC 2013) shows that emissions are trending at or above the A1FI (high) emissions scenario, and secondly, the differences in relative species vulnerability between the high (A1FI) and low (A2/A1B) emissions scenarios were only slight.

Rather than present the relative vulnerability scores for each species based on 2070, below we highlight and discuss the species assessed as likely to be most impacted based on known or inferred environmental thresholds being exceeded. Thresholds for fishery species in northern Australia are not well known with any certainty for the vast majority of species. Where we do have reasonable information on likely thresholds for species, whether

documented or inferred, we can make comments on the likely time frames for these being exceeded and the likely consequences. This does not mean that there are other species that will not be impacted by climate change to the same or even greater extent. Indeed, the vast majority of species assessed for their vulnerability in 2070 were assessed as having "Physiological thresholds unknown". It should also be noted that the likely impacts presented on species by 2030 (Table 8.33) continue to be relevant and are likely to be heightened given continuing trends in the projections for climate variables. There were two species in north-western Australia that were assessed to exceed thresholds by 2070, three in the Gulf of Carpentaria, and seven on the east coast (a total of seven different species overall).

8.5.4.1 Sea cucumbers (sandfish, black teatfish, white teatfish)

All three species of sea cucumber were assessed to exceed particular thresholds by 2070. The key impact on all species is likely to be due to a decrease in pH, which will compromise larval development and survival with potentially catastrophic impacts on populations. Increasing SST will likely cause the spawning season of sandfish to increase, however any benefit of this may be counteracted by a decrease in pH. The effects of ocean acidification on all life history stages of sea cucumber species are highly uncertain however, and are an area of high priority research. The projected increases in SST by 2070 mean that black teatfish populations are not likely to be viable in far northern areas, such as Torres Strait and the far northern Great Barrier Reef, because they prefer shallow water and are a winter spawning species. Black and white teatfish are dependent on coral reef habitats while sandfish are dependent on seagrass meadows. These two habitats are projected to be degraded by 2070 with unknown, but likely negative, consequences for populations of these and other sea cucumber species.

8.5.4.2 Coral trout

The assessments for coral trout are based on knowledge of thresholds for *P. leopardus*, the most common and widespread coral trout species across northern Australia. By 2070 the projected increase in SST will have deleterious effects on the early life history stages. Temperatures above 28 °C will reduce fertilisation rates, hatching rates, larval feeding and development rates, and ultimately will reduce larval survival (Pratchett et al. 2013; see coral trout review in Part 2 companion report). Further, northern populations are as sensitive to thermal changes as southern populations so impacts will be seen first in populations in north-western Australia, the Gulf of Carpentaria, Torres Strait and the northern GBR (Pratchett et al. 2013). Coral trout populations in northern Australia will need to spawn earlier in the season to correspond with SST regimes that are suitable for successful larval survival or there will be much lower population sizes in these regions. It is also possible that by 2070 the negative effects of ocean acidification on juvenile survival demonstrated experimentally (Munday et al. 2012), will manifest. Further, coral reefs are projected to be degraded by 2070 and this will also have negative consequences for post-settlement coral trout.

8.5.4.3 Tropical rock lobster

The main fishery region for tropical rock lobster (TRL) in northern Australia is Torres Strait and in the northern part of he GBR. With recent warmer than usual SST being experienced in several parts of northern Australia (and elsewhere) TRL have been observed to move to deeper cooler parts of coral reefs. It is not well understood what ecological and/or physiological impacts higher SST may have on TRL, however from a fishery perspective the catchability of TRL will be reduced with impacts on fishers. By 2070 ocean acidification and local currents in the north-western Coral Sea could have serious ecological consequences for TRL however, as stated earlier, these are poorly understood in relation to TRL.

8.5.4.4 Saucer scallops

Saucer scallops are taken in only a narrow latitudinal range in southeast Queensland and by 2070 increasing SST are likely to have exceeded their tolerance level, especially since they spawn at the cooler time of year. It may be that scallop populations will shift their range poleward into NSW, or into deeper waters, however this will depend on the availability of local habitat and possibly hydrological characteristics, which appears may also affect successful recruitment.

8.5.4.5 Red throat emperor

Red throat emperor, like saucer scallops, has a narrow latitudinal range on the GBR and SST has been shown to influence movement and catchability (Tobin et al. 2010). Given that they have a preference for coral reefs it is highly possible that increases in SST by 2070 will result in less of a poleward range shift but more of a shift into deeper waters adjacent to the GBR, a habitat they are currently known to occupy. As a consequence of this it is unlikely that there would still be a targeted fishery for this species on the GBR, unless modifications to fishing operations are made.

The project has identified the fishery species of northern Australia that are likely to be most vulnerable to climate change, documented the likely impacts on the key species, and has analysed the reasons for their vulnerability. This provides the basis for what actions are appropriate for these species, whether they be filling information gaps through research, or for industry and/or managers to take action in preparation for the likely changes to fishery species (adaptation options; Section 8.6.2). Further, the project has prioritised species so that the next steps focus on species that are not only likely to experience impacts from climate change but also those that represent the most important socially and/or economically to the people of the different regions in northern Australia.

8.6 Identifying adaptation options

8.6.1 Fisher observations

During the adaptation workshops held with fishing industry members in Darwin and Townsville, we asked participants to convey any changes they had observed over time that related to specific species or to the environment in general. The results of this are given in Tables 8.34 and 8.35 for Darwin and Townsville respectively. The observations from the Darwin workshop relate to Northern Territory including the Gulf of Carpentaria, while observations from the Townsville workshop relate to the east coast.

In general, many of the observations noted are known about but are not well documented, if at all. This is often the case with anecdotal information however in some cases they provide examples of potential impacts, previously documented in this report, that may occur by 2030 or may already be occurring. Without robust monitoring in place, anecdotal reporting is potentially one of the key early warnings for the detection of impacts on fisheries from climate change (or other causes). Stakeholder surveys into the future are likely to be a cost-effective and informative fisheries monitoring tool and should be encouraged.

Table 8.34 Observed fishery changes identified by fishery stakeholders at the Darwin workshop and relating to the Northern Territory, including the Gulf of Carpentaria.

Observation	Attributed cause	Knowledge of observation
In the past 10-20 years there has been increasing saltwater inundation of freshwater floodplains	Sea-level rise	Known. Documented
In some years (more recently) black jewfish spawn in the middle of the year; something very unusual historically	Unknown	Known? Not documented
Poor years for barramundi and mud crab can be attributed to years of lower rainfall	Lower rainfall	Well documented for both species
This year (2013) was a poor banana prawn season	Poor wet season	Relationship well documented
Species preferences for golden snapper and black jewfish commercial fishing has shifted from golden snapper to jewfish	Less golden snapper & increased value for black jewfish (e.g. selling swim bladders)	Known. Not well documented
Fewer sandfish	Too much fishing effort	Not documented. No surveys but shifts in spatial effort in commercial fishing observed

Table 8.35 Observed fishery changes identified by fishery stakeholders at the Townsville workshop and relating to the east coast.

Observation	Attributed cause	Knowledge of observation
Destruction of habitats in recent years (mangroves/seagrass inshore, coral reef shoals offshore). Will decrease tiger prawn/mud crab/red emperor recruitment potentially for many years	Cyclones	Habitat damage is known and documented; impacts have not unequivocally been established
In recent years water temperatures inshore have been too high for large Spanish mackerel	Higher SST	Not known and not yet established
Mackerel, grunters and red snapper are spawning earlier than they used to	Higher SST	Not documented
Baitfish in the southern Great Barrier Reef are less abundant than they used to be	Unknown	Not documented
A lack of longtail tuna coming inshore in recent years	Unknown	Known. Not documented
This year juvenile billfish appeared in Bowling Green Bay in August; earlier than usual and the most in 7 years	Unknown	Known. Not documented

8.6.2   Adaptation options to future scenarios

In the workshop session's stakeholders identified a range of adaptation options based on future impact scenarios derived from the species reviews, data analyses and vulnerability assessments (see Table 8.33 for a summary of potential impacts). These adaptation options were listed as either autonomous or planned. Autonomous adaptation options are changes made by industry based on changed situations within their relevant industry space and are a reactive response to change. For example, with changes in the timing of a target species aggregating on fishing grounds fishers will adapt simply by targeting the fish when they do arrive at the fishing grounds; they change their fishing practices as necessary. Planned adaptation is a more deliberate action taken with pre-planning based on an awareness or anticipation of changing conditions and can include policy changes, business restructuring, altered fishing practices/gear, fish stocking or habitat restoration (e.g. Creighton et al 2013). For each of the adaptation options identified, stakeholders were also asked to consider the potential barriers or challenges to implementing each option. The complete tables of

potential impact scenarios and the adaptation options for the Darwin and Townsville workshops are provided in Appendix 8 with summaries presented below.

The types of autonomous adaptation options identified from both workshops are given in Table 8.36. The list of autonomous adaptation options are the types of actions that fishers already take as part of their routine fishing activities depending on season, species availability/catch rates, weather, and market prices. Both the commercial and recreational fishing sectors have always used these options, however, for the commercial sector there is an underlying economic cost factor in their decision-making. In making these decisions, the commercial sector relies on flexibility in regulatory arrangements to access multiple fisheries and/or species, while the marketability of product is also important since prices paid for product can be variable.

Table 8.36 Summary of the types of autonomous adaptation options identified by stakeholders at both workshops. NB. Some of these options can be both autonomous and planned depending on species and fishery characteristics.

Autonomous adaptation options
Changing the level of effort (increase or decrease catch)(commercial)
Change target species (recreational & commercial)
Change spatial dynamics of fishing effort (commercial)
Change the temporal dynamics of fishing (commercial & recreational)
Increase the level of catch and release (recreational)
Change fishing techniques (e.g. from net to line (commercial); use circle hooks (recreational))
Diversify marketed catch (commercial)
Move to other fisheries (commercial)

The types of planned adaptation options and barriers identified from both workshops, and the entities responsible for their adoption, are summarised in Table 8.37. The range of planned adaptation options identified were placed into four groups: *Alteration of fishing operations*, *Management-based options*, *Research and Development*, and *Looking for alternatives* (Table 8.37). Adaptation options grouped under 'Alteration of fishing operations' were generally adaptation options for industry, although government support in some instances may be needed, and some of which may be autonomous depending on the nature. Examples include more targeting of banana prawns to offset the likely reduced abundance of tiger prawns due to the degradation of seagrass habitat which is critical for juvenile survival. For other species likely to experience population declines adaptation options included: reduced target effort and diversification of target species and fishing locations (e.g. golden snapper, king threadfin, barramundi, coral trout, Spanish mackerel),

and changes in gears and/or fuel types to cheaper alternatives (e.g. biofuels) to reduce running costs (Table 8.37 and Appendix 14.8). The consistent potential barrier identified to these options was the cost involved that is incurred by the fisher. These could be increased fuel costs for travelling further, lower value of alternate species, or costs of changing gears and other operational equipment.

'Management-based options' included the greatest number of adaptation options identified than any other group, highlighting that many options may need to involve regulatory changes and/or policy decision-making. Some of these options involve a review of existing management or the potential introduction of new more flexible arrangements. They included fishery input and output controls, however also included management of land-based influences (river flow), resource allocation among the different fishing sectors (principally recreational and commercial), and development of fishery harvest strategies. For example, in response to future declines in target species population size, adaptation options included: the introduction of catch limits (e.g. king threadfin, barramundi), introduce spawning closures (spatial/temporal; e.g. Spanish mackerel), resource allocation initiatives (e.g. grey mackerel, coral trout, golden snapper), and licence buyouts (e.g. east coast Spanish mackerel) (Table 8.37 and Appendix 14.8). An important option identified in response to likely declining inshore populations due to lower river flows, especially for barramundi, was the improved management of river flows to better balance the needs of agriculture, human consumption and maintenance of ecological processes. The key barriers identified for this group of options were political opposition, cost and bureaucracy. Not surprisingly, the key responsibility in implementing these types of adaptation options lies with government, acknowledging the need for relevant stakeholder involvement (Table 8.37).

Adaptation options grouped under 'Research and Development' included research, monitoring and education, as well as development ideas such as development of Codes of Conduct, improving markets for alternate species and improving product quality. For example, a commercial fisher in one of the workshops produced a dried product not currently marketed being dried wings of queenfish. These are caught as a by-product in inshore net fisheries and occasionally in large numbers. This is a good example of potential value-adding to what is normally a low value product and providing a potential alternative to species currently targeted. The main potential barrier was again perceived to be cost. Although some of these types of options can be industry lead, most would need to involve government in some capacity, particularly as a source of funds, although some may be self-funded options.

The final group of adaptations was 'Looking for alternatives' and included options such as restocking and aquaculture, as well as artificial reefs and habitat restoration. For example, one of the adaptation options identified for tiger prawns was habitat restoration given their

reliance on seagrass meadows for successful juvenile survival. Creighton et al (2013) states that investment into habitat restoration in the Great Barrier Reef region would provide economic benefit from prawn catches (tiger and banana) of at least $45m per annum (post-2018) with a break-even point of less than two years. The key potential barriers identified by stakeholders were cost and political opposition and they all required government to play a key role in their being implemented, with relevant stakeholder involvement (Table 8.37).

It is not surprising that costs were identified as they major barrier to the effective implementation of the identified adaptation options, regardless of their type. The other barriers perceived to be important were political opposition and bureaucracy. Although these are less tangible types of barriers to change, they were the most prominent in the view of the stakeholders present during the workshops. They are therefore very real barriers that need to be addressed. With the development of business case examples such as that described for tiger prawn by Creighton et al (2013), these types of barriers can be more readily overcome.

With limited resources, any future efforts to put adaptation actions into play should involve a process to prioritise actions, but in a transparent way involving relevant stakeholders. Although individuals can lead adaptation actions in response to changes or anticipated changes that affect fisheries, it is clear that government needs to play a key role in facilitating the implementation of actions that fishers, particularly commercial fishers, can adopt. This is because most of the adaptation options identified in the workshops involve regulatory changes, thereby requiring political will and support. Such changes would also require flexible, responsive and adaptive management systems. Clearly, governments will need to play a lead role in climate change adaptation for fisheries in northern Australia.

Given the vast area of northern Australia and the large number of fishery species involved, we have necessarily taken a broad approach in identifying priority species for future attention and the likely climate change impacts on those species. Therefore, the potential adaptation options presented here based on these species and the likely impacts, although identified by the fishery stakeholders present at our workshops (fishery managers, conservation managers, commercial and recreational fishers, and scientists), are also broad in scope and detail. Further, despite industry bodies having good representation, there was a lack of presence from commercial fishers at the adaptation workshops; a key stakeholder group for such workshops. To further develop adaptation options we suggest the need for a regional focus with strong representation of all relevant stakeholder groups and multiple workshops that consider: priority species and likely impacts identified in this project (as well as the underlying mechanisms behind the impacts), and current management and government policy. There is also a need to rigorously prioritise adaptation options, identify complementarity among regions and species, and to identify clear pathways for adoption. Building a solid business case for each option that articulates costs and tangible benefits will

maximise the likelihood of the commitment of the associated resources required for successful adoption.

Table 8.37 Summary of the *types* of planned adaptation options identified by stakeholders from both workshops. Barriers for each adaptation option type are given, and who is responsible for the auctioning of options. The main fishery sector that the adaptation option applies to is given in parentheses.

	Planned adaptation options	Barriers	Responsibility
Alteration of fishing operations	Change target species (commercial)	Cost, public perception	Industry
	Change spatial dynamics of fishing effort (commercial)	Cost	Industry
	Move to other fisheries (commercial)	Cost	Industry
	Change or modify gear types (commercial)	Cost	Industry, Government
	Reduce operating costs, e.g. biofuels	??	Industry, Government
Management-based options	Protection of critical habitats	Cost	Government, stakeholders
	Introduce/revise catch limits (quota – commercial; bag limits – recreational)	Political opposition, bureaucracy, lack of data, cost to commercial sector	Government, stakeholders
	Introduce more flexible management systems	Political opposition, bureaucracy	Government
	Resource allocation (commercial & recreational)	Political opposition, inter-sector conflict, bureaucracy	Government, stakeholders
	Introduce/revise size limits (commercial & recreational)	Political opposition, stakeholder opposition	Government, stakeholders
	Introduce/revise spatial & temporal closures (commercial & recreational)	Political opposition, stakeholder opposition, bureaucracy	Government, stakeholders
	Better management of land-based water for optimal river flows (commercial & recreational)	Competition for water (e.g. agriculture), cost, political opposition, bureaucracy	Government, stakeholders
	Introduce gear restrictions/modifications for specific species (commercial)	Inter-sector conflict, cost	Government, stakeholders
	Develop harvest strategies for all fisheries (commercial & recreational)	Lack of data, lack of education for consumers, bureaucracy	Government, stakeholders
	Government support for effort reduction strategies (e.g. licence buyouts) (commercial)	Political opposition, cost	Government, industry

	Adaptation options	Barriers	Responsibility
Research and Development	Targeted research to inform future fishing levels (commercial & recreational)	Cost	Government, stakeholders
	Better education about fish stocks and regulatory mechanisms (commercial & recreational)	Cost, Political opposition	Government, stakeholders
	Develop and implement Codes of Conduct (commercial & recreational)	??	Government, stakeholders
	Improved product handling and grading standards (commercial)	??	Industry
	Educate and promote targeting of alternate species (commercial & recreational)	Cheap imports	Government, stakeholders
	Improve marketing of target and non-target species to maximise value (commercial)	Cheap imports, marketing costs, oversupply/product value	Industry
	Introduce/maintain fisheries monitoring	Cost, bureaucracy, political opposition	Government
Looking for alternatives	Translocation of mature fish (commercial & recreational)	Cost	Government, stakeholders
	Restocking (commercial & recreational)	Disease/genetic risk, political opposition, costs, technical knowledge	Government, stakeholders
	Provide infrastructure for increased access to fishing areas (commercial & recreational)	Cost, land ownership, political opposition	Government, stakeholders
	Develop aquaculture	Cost	Government, stakeholders
	Introduce artificial habitats (recreational)	Cost, political opposition, green group opposition	Government, stakeholders
	Habitat restoration (commercial & recreational)	Cost	Government, stakeholders

9   BENEFITS AND ADOPTION

This project has taken a deliberate structured and logical approach to inform managers and other fishery stakeholders on appropriate responses to climate change for fisheries in northern Australia. This report provides a concise compendium of all this information relevant to northern Australia and presents this information based on three regional areas: north-western Australia, the Gulf of Carpentaria, and the east coast. Although these regions are expansive and variable in their conditions, they nevertheless ensure relevance for the respective species assessed and the local climate changes expected to occur.

The information generated during this project can be described as discrete outputs each helping to provide the basis for decision-making from fishery and conservation managers, as well as fishers themselves, particularly the commercial sector and related industries who rely on fisheries for their income. A summary of these outputs are given below:

1.   Comprehensive fishery species lists for each of the three regions of northern Australia determined through consultation with fishing stakeholders.

2.   A summary of the observed and projected climate for the three regions of northern Australia using key climate variables relevant to marine fisheries and supporting habitats, and using the most recent information. This information provides an important baseline from which to assess potential impacts (positive or negative) on species in the future.

3.   A review of the key habitats important for marine fishery species and how they are likely to be affected by climate change across each of the three regions.

4.   A comprehensive compendium of reviews for 23 key fisheries species/species groups important for northern Australian fisheries. These include 8 invertebrate species/species groups and 15 fish and shark species, and describe the main fisheries, the species life cycle characteristics and known and inferred information on the sensitivity and response of species to changes in climate variables.

5.   Relative vulnerability assessments carried out for a total of 36 key northern Australian fishery species to climate change by 2030. This is a medium-term time frame that has relevance to stakeholder and management planning horizons. The vulnerability assessment results identified: (i) species that are highly vulnerable to projected climate change and the reasons why, and (ii) species that, because of their economic and/or social importance, should be given a higher priority for action by managers, industry and future research.

6.   The types of adaptation options as identified by stakeholders and based on the types of changes expected to occur in a range of fishery species.

These outputs all documented here together provide a valuable resource for any stakeholder and provide the following outcomes for fishery managers and key stakeholders:

- *Improved understanding of the climate-related changes predicted for the northern Australian region.* The project has documented the most up-to-date climate projections for northern Australia that are relevant to marine fisheries and supporting habitats.

- *A greater understanding of the potential consequences of climate change on northern Australian fishery habitats.* The current and potential impacts of projected changes in climate variables on key fisheries habitats have been identified based on the best available scientific and local knowledge, and cover the following habitats: coral reefs, mangroves, flood plains, bays and estuaries and seagrass meadows.

- *A greater understanding of the potential consequences of climate change on northern Australian fishery species.* The current and potential impacts of projected changes in climate variables on key fisheries species have been identified based on the best available scientific and local knowledge.

- *A clear understanding of which fisheries and which species are most vulnerable to climate change and the source of vulnerability.* The assessment framework determined the relative vulnerability of key fisheries and identified species that had the highest relative vulnerability and the reasons for this. Furthermore, the framework represents a tool that can be revised and/or modified as new and relevant information becomes available.

- *Prioritisation of fishery species for actions and research based on their vulnerability and level of fishery importance.* Using the correlation between fishery vulnerability and importance, the project provides a tool to assist managers, industry and researchers in determining where best to put resources for futures actions including research.

- *An improved understanding of important information gaps and where future research should be directed.* The species reviews and vulnerability assessment highlighted key knowledge gaps, and in particular, where there is high uncertainty in the input information, thereby enabling prioritisation of future research investment.

- *Identification of the types of adaptation responses that are relevant and appropriate.* Based on the likely potential impacts on key fishery species stakeholders identified a range of potential adaptation options that are relevant to the species and appropriate for the fishery circumstances. They also identified the types of issues that need to be overcome for different options, and therefore the options that are able to be implemented easiest, and who is to be responsible if particular adaptation options are to be actioned.

- *Improved capacity of northern Australian fisheries stakeholders and management to prepare for and respond to potential impacts (positive and negative) of climate change.* Knowledge of the potential impacts on fisheries species/habitats and dependent stakeholders provides information about how livelihoods and fishing practices will be influenced. An improved understanding of climate change and potential impacts is essential to enable proactive planning, and to inform management on the efficacy of current management and potentially where to target management actions under future scenarios. Better planning is facilitated through the identification of the main types of adaptation options and the likely barriers that may impede their implementation.

These project outcomes should be of benefit to all fishery stakeholders in northern Australia and provides a strong basis upon which to engage stakeholders about climate change and its implications for fisheries. Accepting and making the changes to adapt to the effects of climate change will be a long and evolving process. The key outputs from this project, particularly the identification of the most vulnerable fisheries to climate change in the respective regions and the causal mechanisms, provide the basis for the process to progress whether through action, education, research, or a combination of these.

Throughout this project several northern Australia fishery stakeholder representatives were involved including fishery and conservation managers, recreational fishers, commercial fishers, and researchers. This involvement has maximised the potential benefits and uptake of project outcomes by the respective groups/agencies.

10 FURTHER DEVELOPMENT

The results of this project have highlighted a number of activities and research priorities for consideration for northern Australian fisheries in the context of a changing climate. Identification of adaptation option types also provides a basis for management and government to begin further discussions and planning with stakeholders for fisheries to be as prepared as possible for climate-related changes to fishery populations and operations. Further development recommendations have been divided into four groups:

Research and monitoring to address key knowledge gaps
- Sea Surface Temperature (SST) was identified as a key environmental variable of significance for northern fishery species yet very little is known of the thermal tolerances of key life history stages of key fishery species. Similar research to that described above for coral trout should be replicated for a number of species notably: tropical rock lobster, Spanish mackerel, tiger prawn, sandfish (and other sea cucumber species), saucer scallops, and red throat emperor.
- Primary productivity changes, linked to rainfall & riverflow but also local hydrodynamics, under future climate scenarios is likely to also significantly affect successful recruitment of fishery species and their prey. Development of models to better understand these processes will be a key piece of research for being able to better predict likely futures for fishery populations in northern Australia. Research that also links habitat repair and re-connectivity with benefits to fisheries productivity will also help to inform these models as well as the efficacy of such strategies as adaptation options.
- Knowledge of the effects of decreasing pH on fishery species is very poor and so comments on the effects of acidification throughout this report are scant. However, given recent studies on coral trout and the effects on juvenile behaviour/olfactory function, this is an area of future research that should be pursued. Given the slow rate of change of pH and the prediction that critical levels are not likely to be reached until at

least 2070, there are other research areas that should be given higher priority. Research should replicate the coral trout work on other key fishery species across a range of taxa, and also include studies on the early development of invertebrates known to have calcium carbonate structures. These should include tropical rock lobster, sandfish (and other sea cucumber species), mud crab and prawn species.

- The continuation of effective and targeted monitoring of key habitats such as coral reefs and sea grass meadows, and the introduction of monitoring mangrove and flood plain habitats to better assess impacts and therefore consequences for important species such as barramundi. Research into the effects of habitat repair on productivity of key fishery species would also help build future efforts that go beyond monitoring.

- The continued efforts by local communities in collecting information on fisheries should be encouraged and supported. One example is the Suntag fish tagging program based near the Rockhampton area over the past ~30 years. Detecting change in fishery species due to long-term climate changes can only be detected by long-term fish data sets such as these.

Governance

- Riverflow to adjacent marine waters has been identified as critical in providing suitable conditions for survival and development of the early life history stages of many northern Australian fishery species. Management of land-based water use while ensuring adequate riverflows for fishery species survival is challenging but likely to be increasingly crucial in a future where rainfall is projected to decrease in many areas, particularly the east coast. Given the many competing uses for water resources (e.g. agriculture) research should be conducted that clearly demonstrates the productivity increases and in particular how this translates to benefits and the extent of these benefits, from adequate water to the environment.

- As with above, the freshwater/estuarine/marine interface represents a critical phase in the life cycle of a plethora of nearshore and offshore fishery species and so healthy waterways will help increase the resilience of fishery species in the future. Therefore, there should be a renewed focus on improving catchment management, e.g. healthy riparian zones, as well as habitat repair and barrier removal from river systems in priority regions.

- To maximize the resilience of key fishery species to climate change and other potential impacts, prudent fisheries management practices that ensures sustainability is essential. Examples of some management measures identified during the vulnerability assessments and species reviews as being needed are: address questions of overfishing for some species, e.g. golden snapper, black jewfish, king threadfin and some sea cucumber species; ensure appropriate management measures are in place for low productivity species (e.g. appropriate minimum size limits to allow maturation and spawning for species such as golden snapper); and maintain and rehabilitate habitats important for sensitive life history stages (e.g. estuaries and sea grass meadows).

- Combining the types of adaptation options identified here with other similar recent studies should be a careful and considered process to best describe these and to take the best approach to facilitate needed actions in the future. This will involve funding and government taking a leading role, particularly for the necessary regulatory/policy changes required to arrive at a management system that is flexible and responsive; a system that would allow management needs and industry needs to respond as needed to the changes predicted. Jurisdictional co-operation will be necessary particularly as species shift.

Extension
- Extension of key aspects of this report to northern Australia fishery regional stakeholders to educate and better inform about potential changes: likely changes in climate; likely impacts on habitats; likely impacts on key fishery species; priority species; potential adaptation options.

Adaptation planning
- To further develop adaptation options we suggest the need for a regional focus with strong representation of all relevant stakeholder groups and multiple workshops that consider: priority species and likely impacts identified in this project (as well as the underlying mechanisms behind the impacts), and current management and government policy. There is also a need to rigorously prioritise adaptation options, identify complementarity among regions and species, and to identify clear pathways for adoption. Building a solid business case for each option that articulates costs and tangible benefits will maximise the likelihood of the commitment of the associated resources required for successful adoption.

Improving future assessments
- This report presents results of an ecological vulnerability assessment; this is just one part of the process. Collection of the relevant information on the adaptive capacity of the fishery (social and economic indicators) would inform the assessment of the *fishery* vulnerability and therefore how fishery participants are best placed to cope with climate change impacts, and to take advantage of opportunities. This would require representatively surveying fishery stakeholders across regions, sectors, particularly commercial, recreational and management. Key fisheries across northern Australia are: inshore fisheries (barramundi), Great Barrier Reef line fishery (coral trout) and Spanish mackerel.
- Ensuring that updated climate modelling information is used in any future climate change assessments for fisheries, particularly where downscaled spatial models have been developed.

Provision of scenario-driven recommendations of adaptive management approaches that provide for the sustainability of northern Australia fisheries in a changing climate.

- The final project workshops worked with stakeholders to identify adaptation options based on likely future fishery scenarios. These options also identified the likely barriers and who is responsible for their implementation and represents an initial, but important, step towards preparing for climate change.

Determination of the vulnerability of northern Australia's fisheries to climate change.

- A key output from the project was the development and application of vulnerability assessments of key fishery species from three key regions of northern Australia. The vulnerability assessments focused on 2030, a medium-term outlook, and one considered to be more relevant to all stakeholders. An assessment was also carried out based on the A1FI emissions scenario for 2070.

Greater understanding of the impacts of short and long term climate variability on northern Australia's key fisheries species, fisheries and regions of northern Australia, and the key environmental drivers. These include identification of priority species, fisheries and/or locations for targeted monitoring.

- The project has delivered as a major output, summary tables of the likely impacts for key northern Australian fishery species and habitats, also identifying the environmental variables of significance. This was done for three regional areas of northern Australia based on emissions scenarios for 2030. The key species likely to be impacted further by changes predicted for 2070 (A1FI emissions scenario) were also identified and the impacts discussed. The vulnerability assessment process also prioritised species for action.

Improved capacity for fisheries management agencies and industry to assess current practices and policies to optimise positioning for future predicted scenarios.

- Collectively, the key outputs of this project provide an informed basis for management and industry to assess current fisheries situations against likely future scenarios. Management as well as commercial and recreational fishing interests were key players during the course of the project having direct input into key outcomes providing a credible base for further extension and uptake by relevant fishery stakeholders.

12 CONCLUSIONS

There are several key conclusions that can be made based on this project. We have grouped these into different categories that generally reflect the different components of the project.

Key species

- Across northern Australia there are many species important to fisheries. The two major species across the entire area are barramundi and mud crab, while other species

were important in some but not all regions: banana and tiger prawns, coral trout, golden snapper and black jewfish, Spanish mackerel and king threadfin.

• The knowledge of how environmental variation affects fishery species is good for 1-2 species, moderate for a few but generally is poor for most species. This means there is certainly scope for sensitivity-based research and the information in this report provides a basis for identifying priority species for such research.

Climate

• Changes in climate across northern Australia will be highly variable depending on the environmental variable and the specific region. The trend is for warmer, less saline and more acidic waters, a rising sea level, stronger cyclones and changed oceanographic conditions not well understood.

• By 2030, north-western Australia will be 0.6 – 0.9 °C warmer, the Gulf of Carpentaria will be 0.3 – 0.6 °C warmer, and both regions will have similar or slightly higher rainfall (0 – 5%)(and riverflow), a sea level rise of between 10 and 20 cm. There will be a weakening of the Leeuwin current on the west coast.

• By 2030, the east coast will be 0.3 – 0.6 °C warmer, will have -10 – 0 % less rainfall (and riverflow), a sea level rise of between 5 and 15 cm and a strengthening of the East Australian Current.

Habitats

• Projected increases in SST will cause more coral bleaching, and ocean acidification will reduce coral growth and structural integrity, resulting in a loss of reef diversity and structure.

• Increased storm severity and extreme riverflow events, resulting in increased turbidity and reduced solar radiation ill reduce seagrass cover and species diversity.

• Sea level rise will cause a landward migration of mangroves and, coupled with altered rainfall patterns, will change the connectivity between rivers and floodplains, resulting in the potential loss of freshwater floodplains.

Data analyses

• Analyses of barramundi CPUE in the Northern Territory provided further evidence of the positive influence of rainfall and riverflow (and floodplain inundation) on barramundi catchability and possibly recruitment.

• In southeast Queensland saucer scallop recruitment is enhanced in years of cooler water. Recruitment also appears to be positively influenced by higher local riverflow and by the presence of a cyclonic current eddy in the Capricorn region.

• Recruitment of Spanish mackerel on the east coast appears to be linked to SST with cooler years positively influencing recruitment, however the causal mechanism is unclear. Analyses did support the hypothesis of a single east coast stock.

Vulnerability and potential impacts

- The project has prioritised species so that the next steps focus on species that are not only likely to experience impacts from climate change but also those that represent the most important socially and/or economically to the people of the different regions in northern Australia.

- Species with the highest ecological vulnerability to climate change tend to have one or more of the following attributes: have an estuarine/nearshore habitat preference during at least part of their life cycle; have low mobility; rely on habitat types predicted to be most impacted by climate change; have low productivity (slow growth/late maturing/low fecundity); are known to be affected by environmental drivers; are fully or overfished.

- Certain species were assessed with a high vulnerability and also high fishery importance and so should be given priority. The highest priority species were **golden snapper, king threadfin, sandfish**, **black teatfish**, **tiger prawn, banana prawn**, **barramundi,** white teatfish and **mangrove jack** (higher priority in bold).

- In the medium-term (2030), the most common impact identified across all species were reduced sizes of populations due mainly to lower rainfall and riverflow which affects primary productivity and therefore survival of early life history stages, and also indirect effects of habitat degradation on key life history stages of certain species. SST is also likely to impact some species by 2030.

- In the longer-term (2070), while changes in rainfall/riverflow, SST and habitat alteration will continue to impact species, ocean acidification and salinity are likely to begin to impact species through disruption of early life history development and habitat effects (particularly coral reefs).

- Rainfall and riverflow are key environmental drivers for fisheries populations in northern Australia through enhancing local primary productivity and larval/juvenile survival, and by connecting key habitats such as estuaries and floodplains. The east coast in particular is a key area for concern due to projected lower rainfall, more extreme (longer) wet and dry periods, coupled with the expected increase in water extraction for land-based use and also having the estuarine habitats modified more than any other region of northern Australia (Creighton et al 2013). Many species use these and nearshore habitats and so are likely to be affected by these changed hydrological conditions, particularly barramundi that use all habitats during all stages of their life history.

- There is a high level of uncertainty in how species, particularly early life history stages, will be affected by changed SST, pH and salinity. However, recent research on coral trout demonstrating behavioural changes that are adverse for survival, suggest there will be surprises in terms of species responses.

Adaptation options

- We were able to group the adaptation options identified by stakeholders into four different types: Alteration of fishing operations, Management-based options, Research and

Development, and Looking for alternatives. Most of the adaptation options identified involved regulatory changes and/or policy decision-making.

• The major barriers to adaptation for northern Australian fisheries were identified as costs, political opposition and bureaucracy. For fisheries to adapt appropriately to climate change all stakeholders will need to play a role, however government will need to need to be a lead player in this process.

Due to the number of fishery species assessed across a vast area, this project took a broad approach to determining the relative vulnerability of key fishery species in northern Australia. Similarly the adaptation options identified by stakeholders were broad in scope and detail. For implementing appropriate adaptation options further, steps need to be considered: further engagement with stakeholders, especially commercial fishers; consideration of alternative options; prioritisation of adaptation responses; and identification and implementation of pathways for successful adoption.

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14 APPENDICES

14.1 Intellectual Property

No patentable or marketable products or processes have arisen from this research. All results will be published in scientific and non-technical literature. The raw data from compulsory fishing logbooks remains the intellectual property of QDAFF and NT Fisheries, whichever is applicable. The raw fishery-independent data from research and monitoring activities remains the intellectual property of QDAFF and NT Fisheries, whichever is applicable. Raw environmental data remains the intellectual property of the relevant agencies: DERM, Bureau of Meteorology, CSIRO, NASA, NOAA, IMOS, DSITIA, Infofish Australia, JCU, DLRM. Intellectual property accruing from analysis and interpretation of raw data vests jointly with JCU, QDAFF, NT Fisheries, Infofish Australia (golden snapper), and the Principal Investigator.

14.2 Staff

Below is a table of staff involved during the course of the project.

Name	Organisation	Funding
David Welch (PI)	C$_2$O Fisheries and JCU, New South Wales	FRDC and In-kind
Julie Robins	QDAFF, Queensland	FRDC and In-kind
Thor Saunders	DoR-Fisheries, Northern Territory	FRDC and In-kind
Richard Saunders	QDAFF @ JCU, Qld	FRDC and In-kind
Andrew Tobin	JCU, Qld	FRDC and In-kind
Alastair Harry	JCU, Qld	FRDC
Colin Simpfendorfer	JCU, Qld	In-kind
Jeffrey Maynard	Maynard Marine, USA	FRDC
Johanna Johnson	C$_2$O Consulting, NSW	FRDC
Gretta Pecl	UTAS, Tasmania	FRDC and In-kind
Bill Sawynok	Infofish Australia, Qld	FRDC
Eric Perez	QSIA, Queensland	FRDC and In-kind
Scott Wiseman	QSIA, Queensland	FRDC and In-kind
Mark Lightowler	QDAFF, Queensland	In-kind
Eddie Jebreen	QDAFF, Queensland	In-kind
John Kung	QDAFF, Queensland	FRDC and In-kind
Randall Owens	GBRMPA, Queensland	In-kind
Darren Cameron	GBRMPA, Queensland	In-kind
Rachel Pears	GBRMPA, Queensland	In-kind
Steve Matthews	DoR-Fisheries, NT	In-kind
Hockseng Lee	DoR-Fisheries, NT	In-kind
Emily Lawson	DoR-Fisheries, NT	FRDC

14.3 1ˢᵗ Workshop participants involved in species identification

Name	Organisation
David Welch (PI)	C$_2$O Fisheries and JCU, New South Wales
Julie Robins	QDAFF, Queensland
Thor Saunders	DoR-Fisheries, Northern Territory
Andrew Tobin	JCU, Qld
Jeffrey Maynard	Maynard Marine, USA
Johanna Johnson	C$_2$O Consulting, NSW
Gretta Pecl	UTAS, Tasmania
Bill Sawynok	Infofish Australia, Qld
Eric Perez	QSIA, Queensland
Mark Lightowler	QDAFF, Queensland
Anthony Roelofs	QDAFF, Queensland
Randall Owens	GBRMPA, Queensland

14.4   Vulnerability assessment workshop participants

Name	Organisation
David Welch (PI)	C_2O Fisheries and JCU, New South Wales
Julie Robins	QDAFF, Queensland
Thor Saunders	DoR-Fisheries, Northern Territory
Andrew Tobin	JCU, Qld
Johanna Johnson	C_2O Consulting, NSW
Gretta Pecl	UTAS, Tasmania
Bill Sawynok	Infofish Australia, Qld
Scott Wiseman	QSIA, Queensland
Eddie Jebreen	QDAFF, Queensland
Richard Saunders	QDAFF, Queensland
Randall Owens	GBRMPA, Queensland
Steve Newman	WA Department of Fisheries

14.5   Adaptation workshop agenda and participants

Darwin agenda

Time	Topic	Speaker(s)
0900	Welcome & purpose of workshop	Thor Saunders
0905	1. Project overview	David Welch
0930	2. Putting climate change into perspective: other key threats and issues across regions and fisheries	Thor Saunders
0940	3. Climate change and fisheries	Gretta Pecl
1000	4. Overview of climate change in north eastern Australia	David Welch
1010	5. Overview and outcomes of vulnerability assessment	David Welch
1035	*Morning tea*	
1100	6. Stakeholder input to finalise VA - questionnaire	All
1130	7. Collating industry perceptions regarding observed oceanographic, ecosystem or fishery changes • Specific species • General	All
1215	8. Climate change impacts on habitats and key species	Thor Saunders
1245	*Lunch*	
1330	9. Eliciting stakeholder responses to species change	David Welch, Gretta Pecl
1340	Exercise: stakeholder responses to species change	All
1515	*Afternoon tea*	
1535	Continued stakeholder exercise	All
1600	Summary of impact responses & follow-up	David Welch
1645	*Close*	

Darwin participants, October 9, 2013
David Welch, JCU/C$_2$O Fisheries (PI)
Thor Saunders, NT Fisheries (CI)
Gretta Pecl, UTas (CI)
Craig Ingram, Amateur Fishermens Association of the Northern Territory
Steve Sly, NT Fisheries (fisheries manager)
Lyn Lambeth, NT Seafood Council
Gilbert Hanson, Northern Land Council

Townsville participants, October 24, 2013
David Welch, JCU/C$_2$O Fisheries (PI)
Thor Saunders, NT Fisheries (CI)
Andrew Tobin, JCU (CI)
Richard Saunders, QDAFF (CI)
Darren Cameron, GBRMPA
Randall Owens, GBRMPA
Bill Sawynok, Infofish Australia
Trevor Fuller, recreational fisher
John Kung, QDAFF
Simon Barry, QDAFF
Scott Wiseman, Queensland Seafood Industry Association
Carolyn Smith-Keune, JCU
Morgan Pratchett, JCU
Glen Murray, commercial fisher

14.6   Species tables of possible environmental drivers

NB. Shaded cells indicate where documented evidence exists.

BARRAMUNDI	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	1	H
rainfall	1	1	1	1	H
pH	0	0	0	0	L
sea level	1	0	1	0	M
salinity (Sur)	1	1	1	1	H
upwelling	0	0	0	0	L
nutrients	1	1	1	1	H
wind/current	0	0	0	0	L
riverflow	1	1	1	1	H

CORAL TROUT	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	1	H
rainfall	0	0	0	0	L
pH	1	0	0	0	L
sea level	0	0	0	0	L
salinity (Sur)	0	0	0	0	L
upwelling	1	1	1	1	H
nutrients	1	1	0	0	M
wind/current	1	0	1	0	M
riverflow	0	0	0	0	L

BANANA PRAWN	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	0	1	H
rainfall	1	1	1	1	H
pH	0	0	0	0	L
sea level	1	0	1	0	M
salinity (Sur)	1	1	1	1	H
upwelling	0	0	0	0	L
nutrients	1	1	1	1	H
wind/current	1	0	1	1	H
riverflow	1	1	1	1	H

SPANISH MACKEREL	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	1	0	**H**
rainfall	1	1	0	0	**M**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	0	0	**L**
upwelling	1	1	0	0	**M**
nutrients	1	1	0	0	**M**
wind/current	1	0	1	0	**M**
riverflow	1	1	0	0	**M**

GOLDEN SNAPPER	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	0	0	0	**L**
rainfall	1	1	0	0	**M**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	1	0	0	0	**L**
upwelling	0	0	0	0	**L**
nutrients	1	1	0	0	**M**
wind/current	0	0	0	0	**L**
riverflow	1	1	0	0	**M**

BLACK JEWFISH	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	0	0	0	**L**
rainfall	1	1	0	0	**M**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	1	0	0	0	**L**
upwelling	0	0	0	0	**L**
nutrients	1	1	0	0	**M**
wind/current	0	0	0	0	**L**
riverflow	1	1	0	0	**M**

REDSPOT KING PRAWN	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	0	0	**M**
rainfall	0	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	0	0	**L**
upwelling	1	1	0	0	**M**
nutrients	1	1	0	0	**M**
wind/current	0	0	0	0	**L**
riverflow	0	0	0	0	**L**

TROPICAL ROCK LOBSTER	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	1	0	**H**
rainfall	0	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	1	1	0	0	**M**
upwelling	1	1	0	0	**M**
nutrients	1	1	0	0	**M**
wind/current	1	0	1	0	**M**
riverflow	0	0	0	0	**L**

TIGER PRAWN	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	1	0	**H**
rainfall	1	0	0	1	**M**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	1	1	1	1	**H**
upwelling	0	0	0	0	**L**
nutrients	0	1	0	0	**L**
wind/current	1	0	0	0	**L**
riverflow	1	1	0	1	**H**

EASTERN KING PRAWN	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	0	0	**M**
rainfall	1	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	1	1	0	1	**H**
upwelling	0	0	0	0	**L**
nutrients	0	0	0	0	**L**
wind/current	1	0	1	0	**M**
riverflow	1	0	0	1	**M**

SAUCER SCALLOP	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	0	0	**M**
rainfall	0	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	0	0	**L**
upwelling	0	0	0	0	**L**
nutrients	0	1	0	0	**L**
wind/current	1	0	1	0	**M**
riverflow	0	0	0	0	**L**

SANDFISH	Recruitment	Growth	Distribution	Catchability	Impact
SST	1	1	1	0	**M**
rainfall	0	0	0	0	**L**
pH	1	1	0	0	**M**
sea level	0	0	1	0	**L**
salinity (Sur)	1	1	1	0	**H**
upwelling	0	0	0	0	**L**
nutrients	1	1	1	0	**H**
wind/current	1	0	0	0	**L**
riverflow	0	0	0	0	**L**

BLACKTIP SHARKS	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	0	**M**
rainfall	0	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	1	0	**L**
upwelling	0	0	0	0	**L**
nutrients	0	0	0	0	**L**
wind/current	0	0	0	0	**L**
riverflow	0	0	1	0	**L**

SCALLOPED HAMMERHEAD SHARK	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	0	**M**
rainfall	0	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	0	0	**L**
upwelling	0	0	1	0	**L**
nutrients	0	0	0	0	**L**
wind/current	0	0	0	0	**L**
riverflow	0	0	0	0	**L**

KING THREADFIN	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	0	**M**
rainfall	1	1	0	0	**M**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	1	0	**L**
upwelling	0	0	0	0	**L**
nutrients	1	1	0	0	**M**
wind/current	0	0	0	0	**L**
riverflow	1	0	0	1	**M**

GOLD BAND SNAPPER	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	0	0	**L**
rainfall	0	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	0	0	**L**
upwelling	1	1	0	0	**M**
nutrients	1	1	0	0	**M**
wind/current	1	0	0	0	**L**
riverflow	0	0	0	0	**L**

MUD CRAB	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	1	**H**
rainfall	1	1	1	1	**H**
pH	0	0	0	0	**L**
sea level	1	0	1	0	**M**
salinity (Sur)	1	1	1	1	**H**
upwelling	1	1	0	0	**M**
nutrients	1	1	1	0	**H**
wind/current	1	0	1	0	**M**
riverflow	1	1	1	1	**H**

GREY MACKEREL	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	1	**H**
rainfall	1	1	0	0	**M**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	1	1	0	0	**M**
upwelling	0	0	0	0	**L**
nutrients	1	1	1	0	**H**
wind/current	0	0	0	0	**L**
riverflow	1	1	0	0	**M**

RED THROAT EMPEROR	Recruitment	Growth	Distribution	Catchability	Impact
SST	0	1	1	1	**H**
rainfall	0	0	0	0	**L**
pH	0	0	0	0	**L**
sea level	0	0	0	0	**L**
salinity (Sur)	0	0	0	0	**L**
upwelling	1	0	1	1	**H**
nutrients	1	1	0	0	**M**
wind/current	0	0	1	0	**L**
riverflow	0	0	0	0	**L**

14.7 Full vulnerability assessment scores (2030, A2/A1FI)

North-western Australia

	Exposure								
	SST+	Altered rainfall	pH decline	Salinity changes	Habitat changes (e.g. loss of productivity, structure or function)	Altered wind/ currents	More severe cyclones/ storms	More extreme riverflow	Exposure index
Golden snapper	3	3	3	3	2	2	2	3	2.63
King threadfin	3	3	3	3	2	2	2	3	2.63
Sand fish	3	2	3	3	3	2	3	1	2.50
Mangrove jack	2	3	3	3	2	2	2	3	2.50
Bull shark	2	3	2	2	1	1	2	3	2.00
Black jew	2	3	3	3	1	2	2	3	2.38
Crimson snapper	1	2	2	2	3	2	1	1	1.75
Saddle tail snapper	1	2	2	2	3	2	1	1	1.75
Pigeye shark	2	2	2	2	1	1	2	2	1.75
Red emperor	1	1	2	2	1	2	1	1	1.38
Coral trout	3	1	2	2	3	2	2	1	2.00
Barramundi	3	3	3	3	2	1	2	3	2.50
Mud crab	3	3	3	3	2	2	2	3	2.63
Grey mackerel	3	2	3	3	2	2	2	2	2.38
Barred javelin	3	2	3	3	2	2	2	2	2.38
Blue threadfin	3	3	3	3	2	2	2	3	2.63
Grass emperor	3	2	3	3	2	2	2	2	2.38
Goldband snapper	1	1	2	2	1	2	1	1	1.38
Scalloped hammerhead	1	1	2	2	1	1	1	1	1.25
Spot tail shark	2	1	2	2	1	1	1	1	1.38
Spanish mackerel	2	3	2	2	2	2	2	3	2.25
Blacktip shark (limbatus)	1	1	2	2	1	1	1	1	1.25
Billfish (Sailfish)	1	1	1	1	1	3	1	1	1.25

	Sensitivity								
	Fecundity (egg production)	Average age at maturity	Generalist v specialist (food & habitat)	Early development duration	Physiological tolerance of stock	Capacity for larvae to disperse	Reliance on environmental drivers (for spawning, settlement)	Potential for timing mismatch (duration of spawning, breeding, moulting)	Sensitivity index
Golden snapper	1	2	2	2	1	2	1	1	1.50
King threadfin	1	2	2	2	1	1	3	2	1.75
Sand fish	1	2	2	2	2	2	2	1	1.75
Mangrove jack	1	2	2	2	1	2	2	2	1.75
Bull shark	3	2	2	3	1	1	2	2	2.00
Black jew	1	1	2	2	1	2	2	1	1.50
Crimson snapper	1	2	2	2	1	2	1	1	1.50
Saddle tail snapper	1	2	2	2	1	2	1	1	1.50
Pigeye shark	3	2	2	3	1	1	2	2	2.00
Red emperor	1	2	1	2	1	3	1	1	1.50
Coral trout	1	1	2	2	1	2	2	2	1.63
Barramundi	1	1	2	2	1	1	3	2	1.63
Mud crab	1	1	2	2	1	3	3	2	1.88
Grey mackerel	1	1	2	2	1	2	2	1	1.50
Barred javelin	1	1	2	2	1	2	2	1	1.50
Blue threadfin	1	1	2	2	1	1	1	1	1.25
Grass emperor	1	1	2	2	1	2	1	1	1.38
Goldband snapper	1	2	2	2	1	2	1	1	1.50
Scalloped hammerhead	3	3	1	3	1	1	2	3	2.13
Spot tail shark	3	2	2	3	1	1	1	2	1.88
Spanish mackerel	1	1	2	2	1	2	2	1	1.50
Blacktip shark (limbatus)	3	2	1	3	1	1	1	3	1.88
Billfish (Sailfish)	1	2	2	2	1	3	2	1	1.75

	Adaptive Capacity							
	Stock status	Replenishment potential	Suitable alternate habitat availability	Species mobility	Non-fishing pressures on stock	*Adaptive Capacity index*	AC normalisation	1-AC
Golden snapper	1	1	2	2	2	1.60	0.57	0.43
King threadfin	1	2	2	2	2	1.80	0.64	0.36
Sand fish	2	2	2	1	2	1.80	0.64	0.36
Mangrove jack	2	1	3	2	2	2.00	0.71	0.29
Bull shark	2	1	2	3	3	2.20	0.79	0.21
Black jew	1	3	3	2	2	2.20	0.79	0.21
Crimson snapper	3	1	2	2	2	2.00	0.71	0.29
Saddle tail snapper	3	1	2	2	2	2.00	0.71	0.29
Pigeye shark	2	1	2	3	3	2.20	0.79	0.21
Red emperor	2	1	2	2	2	1.80	0.64	0.36
Coral trout	3	3	2	2	2	2.40	0.86	0.14
Barramundi	3	2	3	2	2	2.40	0.86	0.14
Mud crab	3	3	3	1	2	2.40	0.86	0.14
Grey mackerel	3	3	2	2	2	2.40	0.86	0.14
Barred javelin	2	3	3	2	2	2.40	0.86	0.14
Blue threadfin	3	3	2	2	2	2.40	0.86	0.14
Grass emperor	2	3	3	2	2	2.40	0.86	0.14
Goldband snapper	3	1	2	2	3	2.20	0.79	0.21
Scalloped hammerhead	3	1	3	3	2	2.40	0.86	0.14
Spot tail shark	3	1	2	3	3	2.40	0.86	0.14
Spanish mackerel	3	3	2	3	2	2.60	0.93	0.07
Blacktip shark (limbatus)	3	1	3	3	3	2.60	0.93	0.07
Billfish (Sailfish)	2	3	3	3	3	2.80	1.00	0.00

	Potential Impacts (PI) (negative)		
	PI = E * S	Direction of impact	PI Index
Golden snapper	3.94	0	3.94
King threadfin	4.59	0	4.59
Sand fish	4.38	0	4.38
Mangrove jack	4.38	0	4.38
Bull shark	4.00	0	4.00
Black jew	3.56	0	3.56
Crimson snapper	2.63	0	2.63
Saddle tail snapper	2.63	0	2.63
Pigeye shark	3.50	0	3.50
Red emperor	2.06	0	2.06
Coral trout	3.25	1	4.25
Barramundi	4.06	0	4.06
Mud crab	4.92	-1	3.92
Grey mackerel	3.56	0	3.56
Barred javelin	3.56	0	3.56
Blue threadfin	3.28	0	3.28
Grass emperor	3.27	0	3.27
Goldband snapper	2.06	0	2.06
Scalloped hammerhead	2.66	0	2.66
Spot tail shark	2.58	0	2.58
Spanish mackerel	3.38	0	3.38
Blacktip shark (limbatus)	2.34	0	2.34
Billfish (Sailfish)	2.19	0	2.19

Gulf of Carpentaria

	Exposure								
	SST+	Altered rainfall	pH decline	Salinity changes	Habitat changes (e.g. loss of productivity, structure or function)	Altered wind/ currents	More severe storms	More extreme riverflow	*Exposure index*
Golden snapper	3	3	3	3	2	2	2	3	2.63
King threadfin	3	3	3	3	2	2	2	3	2.63
Sand fish	3	2	3	3	3	2	3	1	2.50
Tiger prawn (esculentus)	3	3	3	3	3	2	3	3	2.88
Mangrove jack	2	3	3	3	2	2	2	3	2.50
Banana prawn	3	3	3	3	2	1	3	3	2.63
Tropical lobster	3	1	2	2	3	3	2	1	2.13
Black jew	2	3	3	3	1	2	2	3	2.38
Pigeye shark	2	2	2	2	1	1	2	2	1.75
Red emperor	1	1	2	2	1	2	1	1	1.38
Barramundi	3	3	3	3	2	1	2	3	2.50
Mud crab	3	3	3	3	2	2	2	3	2.63
Grey mackerel	3	2	3	3	2	2	2	2	2.38
Barred javelin	3	2	3	3	2	2	2	2	2.38
Blue threadfin	3	3	3	3	2	2	2	3	2.63
Coral trout	3	1	2	2	3	2	2	1	2.00
Scalloped hammerhead	1	1	2	2	1	1	1	1	1.25
Spot tail shark	2	1	2	2	1	1	1	1	1.38
Spanish mackerel	2	3	2	2	2	2	2	3	2.25
Blacktip shark (limbatus)	1	1	2	2	1	1	1	1	1.25
Billfish (Sailfish)	1	1	1	1	1	3	1	1	1.25

	Sensitivity								
	Fecundity (egg production)	Average age at maturity	Generalist v specialist (food & habitat)	Early development duration	Physiological tolerance of stock	Capacity for larvae to disperse	Reliance on environmental drivers (for spawning, settlement)	Potential for timing mismatch (duration of spawning, breeding, moulting)	*Sensitivity index*
Golden snapper	1	2	2	2	1	2	1	1	1.50
King threadfin	1	2	2	2	1	1	3	2	1.75
Sand fish	1	2	2	2	2	2	2	1	1.75
Tiger prawn (esculentus)	1	1	2	2	2	1	2	1	1.50
Mangrove jack	1	2	2	2	1	2	2	2	1.75
Banana prawn	1	1	2	2	1	1	3	3	1.75
Tropical lobster	2	2	2	1	3	3	3	1	2.13
Black jew	1	1	2	2	1	2	2	1	1.50
Pigeye shark	3	2	2	3	1	1	2	2	2.00
Red emperor	1	2	1	2	1	3	1	1	1.50
Barramundi	1	1	2	2	1	1	3	2	1.63
Mud crab	1	1	2	2	1	3	3	2	1.88
Grey mackerel	1	1	2	2	1	2	2	1	1.50
Barred javelin	1	1	2	2	1	2	2	1	1.50
Blue threadfin	1	1	2	2	1	1	1	1	1.25
Coral trout	1	1	2	2	1	2	2	2	1.63
Scalloped hammerhead	3	3	1	3	1	1	2	3	2.13
Spot tail shark	3	2	2	3	1	1	1	2	1.88
Spanish mackerel	1	1	2	2	1	2	2	1	1.50
Blacktip shark (limbatus)	3	2	1	3	1	1	1	3	1.88
Billfish (Sailfish)	1	2	2	2	1	3	2	1	1.75

	Adaptive Capacity							
	Stock status	Replenishment potential	Suitable alternate habitat availability	Species mobility	Non-fishing pressures on stock	*Adaptive Capacity index*	AC normalisation	1-AC
Golden snapper	1	1	2	2	2	1.60	0.57	0.43
King threadfin	1	2	2	2	2	1.80	0.64	0.36
Sand fish	2	2	2	1	2	1.80	0.64	0.36
Tiger prawn (esculentus)	3	3	2	1	1	2.00	0.71	0.29
Mangrove jack	2	1	3	2	2	2.00	0.71	0.29
Banana prawn	3	3	1	1	2	2.00	0.71	0.29
Tropical lobster	3	2	3	2	2	2.40	0.86	0.14
Black jew	1	3	3	2	2	2.20	0.79	0.21
Pigeye shark	2	1	2	3	3	2.20	0.79	0.21
Red emperor	2	1	2	2	2	1.80	0.64	0.36
Barramundi	3	2	3	2	2	2.40	0.86	0.14
Mud crab	3	3	3	1	2	2.40	0.86	0.14
Grey mackerel	3	3	2	2	2	2.40	0.86	0.14
Barred javelin	2	3	3	2	2	2.40	0.86	0.14
Blue threadfin	3	3	2	2	2	2.40	0.86	0.14
Coral trout	3	3	2	2	2	2.40	0.86	0.14
Scalloped hammerhead	3	1	3	3	2	2.40	0.86	0.14
Spot tail shark	3	1	2	3	3	2.40	0.86	0.14
Spanish mackerel	3	3	2	3	2	2.60	0.93	0.07
Blacktip shark (limbatus)	3	1	3	3	3	2.60	0.93	0.07
Billfish (Sailfish)	2	3	3	3	3	2.80	1.00	0.00

	Potential Impacts (PI) (negative)		
	PI = E * S	Direction of impact	PI Index
Golden snapper	3.94	0	3.94
King threadfin	4.59	0	4.59
Sand fish	4.38	0	4.38
Tiger prawn (esculentus)	4.31	1	5.31
Mangrove jack	4.38	0	4.38
Banana prawn	4.59	-1	3.59
Tropical lobster	4.52	1	5.52
Black jew	3.56	0	3.56
Pigeye shark	3.50	0	3.50
Red emperor	2.06	0	2.06
Barramundi	4.06	0	4.06
Mud crab	4.92	-1	3.92
Grey mackerel	3.56	0	3.56
Barred javelin	3.56	0	3.56
Blue threadfin	3.28	0	3.28
Coral trout	3.25	0	3.25
Scalloped hammerhead	2.66	0	2.66
Spot tail shark	2.58	0	2.58
Spanish mackerel	3.38	0	3.38
Blacktip shark (limbatus)	2.34	0	2.34
Billfish (Sailfish)	2.19	0	2.19

East coast

	Exposure								
	SST+	Altered rainfall	pH decline	Salinity changes	Habitat changes (e.g. loss of productivity, structure or function)	Altered wind/ currents	More severe cyclones/ storms	More extreme riverflow; total reduced	*Exposure index*
Black teat fish	3	1	2	2	3	2	3	1	2.13
King threadfin	3	3	3	3	2	2	2	3	2.63
Sand fish	3	2	3	3	3	2	3	1	2.50
Barramundi	3	3	3	3	2	1	2	3	2.50
Tiger prawn (esculentus)	3	3	3	3	3	2	3	3	2.88
Golden snapper	3	3	3	3	2	2	2	3	2.63
White teat fish	2	1	2	2	3	2	3	1	2.00
Banana prawn	3	3	3	3	2	1	3	3	2.63
Mangrove jack	2	3	3	3	2	2	2	3	2.50
Tropical lobster	3	1	2	2	3	3	2	1	2.13
Scallops	1	1	2	2	2	2	1	1	1.50
Red emperor	1	1	2	2	1	2	1	1	1.38
Mud crab	3	3	3	3	2	1	3	3	2.63
Dusky flathead	3	3	3	3	2	2	2	3	2.63
Red throat emperor	1	1	2	2	3	2	2	1	1.75
Coral trout	3	1	2	2	3	2	2	1	2.00
Grey mackerel	3	2	3	3	2	2	2	2	2.38
Barred javelin	3	2	3	3	2	2	2	2	2.38
Red spot king prawn	2	1	2	2	3	2	3	1	2.00
Scalloped hammerhead	1	1	2	2	1	1	1	1	1.25
Spot tail shark	2	1	2	2	1	1	1	1	1.38
Spanish mackerel	2	3	2	2	2	2	2	3	2.25
Blue threadfin	3	3	3	3	2	2	2	3	2.63
Blacktip sharks (limbatus)	1	1	2	2	1	1	1	1	1.25
Eastern king prawn	1	2	2	2	1	3	1	1	1.63
Billfish (black marlin)	1	1	1	1	1	3	1	1	1.25
Spotted mackerel	3	3	3	3	2	2	1	3	2.50
Moreton Bay bug	1	1	2	2	1	3	1	1	1.50

	Sensitivity								
	Fecundity (egg production)	Average age at maturity	Generalist v specialist (food & habitat)	Early development duration	Physiological tolerance of stock	Capacity for larvae to disperse	Reliance on environmental drivers (for spawning, settlement)	Potential for timing mismatch (duration of spawning, breeding, moulting)	Sensitivity index
Black teat fish	1	2	3	2	2	2	1	2	1.88
King threadfin	1	2	2	2	1	1	3	2	1.75
Sand fish	1	2	2	2	2	2	2	1	1.75
Barramundi	1	2	2	2	1	1	3	2	1.75
Tiger prawn (esculentus)	1	1	2	2	1	1	2	1	1.38
Golden snapper	1	2	2	2	1	2	1	1	1.50
White teat fish	1	2	3	2	2	2	1	2	1.88
Banana prawn	1	1	2	2	1	1	3	3	1.75
Mangrove jack	1	2	2	2	1	2	2	2	1.75
Tropical lobster	2	2	2	1	3	3	3	1	2.13
Scallops	1	1	2	2	1	2	3	2	1.75
Red emperor	1	2	1	2	1	3	1	1	1.50
Mud crab	1	1	2	2	1	3	3	2	1.88
Dusky flathead	1	2	2	2	1	2	2	2	1.75
Red throat emperor	1	2	2	2	1	2	2	1	1.63
Coral trout	1	1	2	2	1	2	2	2	1.63
Grey mackerel	1	1	2	2	1	2	2	1	1.50
Barred javelin	1	1	2	2	1	2	2	1	1.50
Red spot king prawn	1	1	2	2	2	2	1	1	1.50
Scalloped hammerhead	3	3	1	3	1	1	2	3	2.13
Spot tail shark	3	2	2	3	1	1	1	2	1.88
Spanish mackerel	1	1	2	2	1	2	2	1	1.50
Blue threadfin	1	1	2	2	1	1	1	1	1.25
Blacktip sharks (limbatus)	3	2	1	3	1	1	1	3	1.88
Eastern king prawn	1	1	1	2	1	3	2	2	1.63
Billfish (black marlin)	1	2	2	2	1	3	3	2	2.00
Spotted mackerel	1	1	2	2	2	2	1	2	1.63
Moreton Bay bug	2	1	2	1	2	3	1	2	1.75

	Adaptive Capacity							
	Stock status	Replenishment potential	Suitable alternate habitat	Species mobility	Non-fishing pressures on stock	*Adaptive Capacity index*	AC normalisation	1-AC
Black teat fish	1	1	2	1	2	1.40	0.50	0.50
King threadfin	1	2	2	2	1	1.60	0.57	0.43
Sand fish	2	2	2	1	2	1.80	0.64	0.36
Barramundi	3	2	2	2	1	2.00	0.71	0.29
Tiger prawn (esculentus)	3	3	2	1	1	2.00	0.71	0.29
Golden snapper	2	1	2	2	2	1.80	0.64	0.36
White teat fish	2	2	2	1	2	1.80	0.64	0.36
Banana prawn	3	3	1	1	2	2.00	0.71	0.29
Mangrove jack	2	1	3	2	2	2.00	0.71	0.29
Tropical lobster	3	2	3	2	2	2.40	0.86	0.14
Scallops	3	3	3	1	1	2.20	0.79	0.21
Red emperor	2	1	2	2	2	1.80	0.64	0.36
Mud crab	3	3	3	1	2	2.40	0.86	0.14
Dusky flathead	3	2	3	2	2	2.40	0.86	0.14
Red throat emperor	3	2	2	2	2	2.20	0.79	0.21
Coral trout	3	3	2	2	2	2.40	0.86	0.14
Grey mackerel	3	3	2	2	2	2.40	0.86	0.14
Barred javelin	2	3	3	2	2	2.40	0.86	0.14
Red spot king prawn	3	3	2	1	3	2.40	0.86	0.14
Scalloped hammerhead	3	1	3	3	2	2.40	0.86	0.14
Spot tail shark	3	1	2	3	3	2.40	0.86	0.14
Spanish mackerel	3	3	2	3	2	2.60	0.93	0.07
Blue threadfin	3	3	2	2	3	2.60	0.93	0.07
Blacktip sharks (limbatus)	3	1	3	3	3	2.60	0.93	0.07
Eastern king prawn	3	3	3	2	3	2.80	1.00	0.00
Billfish (black marlin)	3	2	3	3	3	2.80	1.00	0.00
Spotted mackerel	3	3	3	3	2	2.80	1.00	0.00
Moreton Bay bug	3	3	3	2	3	2.80	1.00	0.00

	Potential Impacts (negative)		
	PI = E * S	Direction of impact	PI Index
Black teat fish	3.98	1	4.98
King threadfin	4.59	1	5.59
Sand fish	4.38	0	4.38
Barramundi	4.38	1	5.38
Tiger prawn (esculentus)	3.95	1	4.95
Golden snapper	3.94	0	3.94
White teat fish	3.75	0	3.75
Banana prawn	4.59	0	4.59
Mangrove jack	4.38	0	4.38
Tropical lobster	4.52	1	5.52
Scallops	2.63	1	3.63
Red emperor	2.06	0	2.06
Mud crab	4.92	0	4.92
Dusky flathead	4.59	0	4.59
Red throat emperor	2.84	0	2.84
Coral trout	3.25	1	4.25
Grey mackerel	3.56	0	3.56
Barred javelin	3.56	0	3.56
Red spot king prawn	3.00	0	3.00
Scalloped hammerhead	2.66	0	2.66
Spot tail shark	2.58	0	2.58
Spanish mackerel	3.38	0	3.38
Blue threadfin	3.28	0	3.28
Blacktip sharks (limbatus)	2.34	0	2.34
Eastern king prawn	2.64	1	3.64
Billfish (black marlin)	2.50	0	2.50
Spotted mackerel	4.06	0	4.06
Moreton Bay bug	2.63	0	2.63

14.8 Raw adaptation option tables

Appendix 14.8.1. Potential adaptation options identified by stakeholders at the adaptation workshop held in Darwin. Stakeholder identified autonomous and planned adaptation options as well as potential barriers to each option. Causal climate factors of the potential impacts are given in parentheses.

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Tiger prawns: Decreased juvenile growth and recruitment (seagrass habitat degradation)	• More targeting of banana prawns	• Habitat restoration	• Cost
Banana prawns: Increased population size (rainfall, riverflow); increased juvenile survival (sea level rise; increased mangrove habitat)	• Increased effort/extend season		
Golden snapper: Reduced spawning biomass (SST)	• Reduce effort, cap catch • Target other species (recreational)	• Target other species (commercial) • Shift to new areas (commercial) • Translocation of mature fish • Release of juveniles (hatchery reared)	• Costs associated with moving (fuel, lack of access and infrastructure) • Cost of translocation
Barramundi: Decreased abundance (rainfall, riverflow, sea level rise, habitat changes)	• Reduce effort • Target other species • Spread effort • Improve value of product	• Increase access (physical access points) to currently available areas • Introduce catch limits • Promote/educate to target other species • Marketing to improve value • Marketing to increase value	• Cost of access and land tenure • Indigenous and pastoral land rights • Political will • Cost of marketing • Cost of advertising • Competing with cheap imports

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Barramundi: increased abundance (rainfall, riverflow, sea level rise)	• Increase effort in 1-3 years time		
King threadfin: Decreased abundance (rainfall, riverflow, sea level rise, habitat changes)	• Reduce effort • Target other species • Spread effort • Improve value of product	• Increase access (physical access points) to currently available areas • Introduce catch limits • Promote/educate to target other species • Marketing to improve value • Promote product to increase value	• Cost of access and land tenure • Indigenous and pastoral land rights • Political will • Cost of marketing • Cost of advertising • Competing with cheap imports
Sandfish: Reduced survival of juveniles (seagrass habitat degradation)	• Reduce effort	• Target new species • Ranching • Habitat restoration • Monitoring	• Gaining access to offshore areas • Concerns over disease risk and genetic risk • Cost of restoring habitat/monitoring • Permits, applications, approvals, trials
Mud crab: Increased abundance (rainfall, riverflow, sea level rise, habitat changes)	• Increase effort	• Improve export market • Reduction in size limit (commercial)	• Cost of marketing • Political will • Regulation change
Mud crab: Increased catchability (SST)	• Increase effort	• Monitor to assess whether is an increase in abundance	• Ensure sustainable fishing

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Black jewfish: Decreased abundance (rainfall, riverflow)	• Reduce effort, cap catch • Target other species (recreational)	• Target other species (commercial) • Shift to new areas (commercial) • Translocation of mature fish • Release of juveniles (hatchery reared)	• Costs associated with moving (fuel, lack of access and infrastructure) • Cost of translocation
Spanish mackerel: Decreased survival of larvae/juveniles (SST)	• Reduce effort	• Change Fisheries • Introduce catch limits • Introduce monitoring	• Cost of monitoring • Changing management
Grey mackerel: Decreased abundance (riverflow, rainfall, SST)	• Reduce effort • Target other species	• Target other species • Change gear types (issues longline catching large sharks)	• Public perception of taking more shark
All species: Higher inter-annual variability in catch rates of all species	• Diversify species and location	• Reduce operating costs, eg. Biofuels • Develop niche markets, eg. Promote local markets to reduce costs such as freight • 'Green' marketing	• Market barriers with taking advantage of 'good years' • Additional marketing costs • Local purchase value

Appendix 14.8.2. Potential adaptation options identified by *recreational fishing* stakeholders at the adaptation workshop held in Townsville. Stakeholder identified autonomous and planned adaptation options as well as potential barriers to each option. Causal climate factors of the potential impacts are given in parentheses.

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Barramundi: Decreased abundance, growth and catchability (rainfall, riverflow, sea level rise, habitat changes)	• Change target species • Increase catch and release • Reduce harvest • Change fishing techniques while still targeting species	• Enhanced restocking rates • Remove barriers to habitats critical in their life cycle (habitat & connectivity restoration) • Improve infrastructure to access other areas where barramundi are caught • Better management of environmental flows from impoundments on rivers • Reduce commercial catch/effort • Restriction of access to currently fishable waters	• Government & public opposition to restocking • Fingerling availability • Genetic integrity of wild stocks • High costs and lack of expertise • Competition for water use • Industry and political opposition
Coral trout: Decreased abundance and catchability (SST, coral reef degradation, cyclones)	• Change target species • Reduce targeted effort	• Introduce artificial habitats	• Government & and green group opposition
Coral trout: Earlier timing of spawning in northern regions (SST)	•	• Change the current spawning closure months	•

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Mud crab: Decreased abundance (rainfall, riverflow, sea level rise)	•	• Establish commercial aquaculture to reduce effort by recreational fishers on wild stocks • Reduce catch/effort (e.g. bag limits, pot limits) • Better management of environmental flows • Preserve mangrove habitats • Introduce closed areas/seasons	• Costs; cannibalism reducing grow-out survival • Political opposition • Cost; politics • Cost • Political opposition
Spanish mackerel: Earlier timing of spawning in the north (SST)	• Target Spanish mackerel earlier in the season	•	•
Spanish mackerel: Southerly range extension (SST)	• New fisheries in northern NSW	• New fisheries in northern NSW	•
Spanish mackerel: Decreased abundance (SST)	• Target alternative species	• Reduce catch (TAC, bag limits) • Introduce flexible management systems • Reduce effort (licenses) • Protect spawning stock (closed season)	• Political opposition • Political opposition • Political opposition • Political opposition

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Grey mackerel: Decreased abundance (SST, rainfall, riverflow)	•	• Ban netting for grey mackerel • Make them a recreational only species • Make them a line-caught only species • Resource allocation among sectors	• Political opposition • Political opposition • Political opposition • Political opposition
King threadfin: Decreased abundance (rainfall, riverflow)	• Change target species • Increase catch and release • Reduce harvest • Change fishing techniques while still targeting species	• Better management of environmental flows from impoundments on rivers • Reduce commercial catch/effort • Restriction of access to currently fishable waters	• High costs and lack of expertise • Competition for water use • Industry and political opposition • Industry and political opposition
Golden snapper: Reduced spawning biomass (SST)	•	• Reduce commercial net effort (deep set gillnets in particular) • Recreational only species • Increase the minimum legal size	•
Tropical rock lobster: Reduced abundance and catchability in northern areas/Increased abundance in southern areas (SST, currents)	• Change target species	• Move southern commercial fishery boundary • Change the commercial TAC • Change recreational catch limits • Contained recreational access on the east coast & Torres Strait	• Bureaucratic processes

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Banana prawns: Decreased abundance (rainfall, riverflow, sea level rise, SST)	• Less targeted effort	• Review restrictions on recreational limits	• Political sensitivity/resistance
Banana prawns: Increased abundance (rainfall, riverflow, sea level rise)	• Increase effort	• Review restrictions on recreational limits	• Political sensitivity/resistance
All species: Higher inter-annual variability in catch rates	• Change target species	• Educate to change values/behaviour based on status of stocks (e.g. release more to maintain larger more fecund fish) • Research to improve recruitment knowledge and predictions of strong year classes which may support increased harvest levels	• Cost and lack of government support • Cost
All species: Changes in the timing of spawning and/or recruitment	•	• Better protection of spawning fish by education and regulatory mechanisms	• Political resistance to regulatory intervention

Appendix 14.8.3. Potential adaptation options identified by *commercial fishing* stakeholders at the adaptation workshop held in <u>Townsville</u>. Stakeholder identified autonomous and planned adaptation options as well as potential barriers to each option. Causal climate factors of the potential impacts are given in parentheses.

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Barramundi: Decreased abundance, growth and catchability (rainfall, riverflow, sea level rise, habitat changes)	• Retain other species • Move to alternate fishing areas	• Through improved marketing increase the value of barramundi and other species • Restocking	• Lack of knowledge of other areas • Additional costs to fishers • Logistical constraints • Hostility
Coral trout: Decreased abundance and catchability (SST, coral reef degradation, cyclones)	• Change fisheries (move into another fishery) • Stop fishing for trout • Target alternate species	• Rotational opening of green zones • Change to other fisheries (flexible licensing) • Improve the value of coral trout/other species (marketing)	• Already high value of coral trout
Coral trout: Earlier timing of spawning in northern regions (SST)	•	• Adjust timing of spawning closure • Possible further closures • Restocking • Resource allocation	• Legislative processes • Lack of information
Mud crab: Decreased abundance (rainfall, riverflow, sea level rise)	• Move to other areas • Change to other fisheries	• Develop a harvest strategy • Develop and implement a Code of Practice • Introduce a levy to support industry quota • Resource allocation	• Legislative processes

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
Mud crab: Increased catchability (SST)	• Catch more crabs • Increase effort	• Introduce fisheries dependent quota • Improve stock assessments • Introduce higher grading standards	• Over supply of product leading to lower prices
Spanish mackerel: Earlier timing of spawning in the north (SST)	• Adjust the timing of peak effort • Switch to targeting other species in the interim	• Resource allocation	• Conflict with other fisheries
Spanish mackerel: Southerly range extension (SST)	• Move fishing operations • Target other species	• Resource allocation	• Conflict with other fisheries
Spanish mackerel: Decreased abundance (SST)	• Reduce effort • Switch target species	• Introduce exit options for the commercial sector (licence buyout) • Apply appropriate quota • Resource allocation	• Lack of funding • Lack of data • Legislative processes
Grey mackerel: Decreased abundance (SST, rainfall, riverflow)	• Reduce effort • Switch target species	• Adapt fishing technology to be line caught species only • Switch targeting to other species • Shorten net shots • Implement a Code of Conduct • Improve product quality control	• Changing fisher behaviour and fishing operations • Lower catch rates • Funding • Legislative processes • Clashes with the recreational sector

Potential Impact	Autonomous adaptation	Potential adaptation actions	Potential barriers
King threadfin: Decreased abundance (rainfall, riverflow)	• Switch to target other species	• Use a smaller net mesh size • Reduce the recreational bag limit • Conduct research for strategies to increase abundance	• Need for specialist nets • Education • Lack of funding
Tiger prawns: Decreased abundance and catchability (seagrass, SST, rainfall)	•	• Bycatch reduction and improvement • Utilisation of bycatch • Marketing of permitted catch • Develop harvest strategies for optimisation of seasons and catch to maximise economic return	• Regulatory impediments • Fishers perception of extra competition by trawl • Lack of labelling requirements • Education of consumers
Eastern king prawns: Decreased abundance (SST, wind/currents)	•	• Bycatch reduction and improvement • Utilisation of bycatch • Marketing of permitted catch • Develop harvest strategies for optimisation of seasons and catch to maximise economic return	• Regulatory impediments • Fishers perception of extra competition by trawl • Lack of labelling requirements • Education of consumers
Banana prawns: Decreased abundance and catchability (rainfall, sea level rise, SST)	•	• Bycatch reduction and improvement • Utilisation of bycatch • Marketing of permitted catch • Develop harvest strategies for optimisation of seasons and catch to maximise economic	• Regulatory impediments • Fishers perception of extra competition by trawl • Lack of labelling requirements • Education of consumers

		return	

14.9 R code for Spanish mackerel CPUE standardisation

lme(log(Wgt)~as.factor(Finyear)+sin(2*pi*Moon)+cos(2*pi*Moon)+sin(4*pi*Moon)+cos(4*pi*Moon)+Grid,data=Catch,random=~1|Fisher,na.action=na.omit)

Wgt: Single day catch of *S. commerson* in kg (multi-day records were removed)

Finyear: Financial year (1st July in calendar year to 30th June in following calendar year)

Moon: Luminosity or lunar phase (calculated from 'phenology' package in R)

Grid: Catch reporting grid

Fisher: Unique vessel ID

14.10 Spanish mackerel age-length key generated age structure

Population age structure of *S. commerson* from four regions off northeastern Australia, 2001- 2011, showing numbers of fish, *N*, sampled of age, *x*. For each year of sampling, *i,* the Studentized residuals from a linear regression of $\log(N_i) = a+bx_i$ were used to provide replicate estimates of relative abundance in the year, *i - x*. Only fish aged 2-11 were included in the analysis since 0-1 year old fish were not fully recruited the fishing gear, and fish > 11 years (excluded below) were relatively rare.

Region	Age	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	Region	2004	2005	2006	2007	2008	2009	2010	2011
Townsville	0								5	1	2		Mackay					1	2		
	1	50	48	1	42	161	34	46	436	143	86	104		10		35	1	61	121	63	40
	2	66	1098	528	473	485	164	296	548	698	749	1084		81	37	69	25	84	431	431	217
	3	197	249	298	422	345	350	178	312	129	593	812		76	62	156	10	37	55	213	130
	4	748	228	42	274	252	189	288	215	90	144	596		23	21	71	4	37	26	40	68
	5	74	405	48	44	116	125	107	223	38	107	128		3	12	37		39	11	19	15
	6	68	36	161	41	35	70	81	119	63	47	106		3	3	16		22	9	19	13
	7	154	100	56	60	19	11	26	116	39	68	49		4		2		12	8	11	6
	8			50	18	29	28	8	65	27	34	54			2	2		9	4	6	7
	9	9		17	16	18	24	7	5	14	26	35				2		1	2	2	3
	10			8	11	17	8	8	7	12	10	15						1		4	1
	11					15	21	3	7	1		5				1		3		1	2
Rockhampton	0								2				South					3			
	1				60	75	49	31	70	32	40	23		20	4	18	11	320	203	58	32
	2			16	327	375	34	92	29	159	82	162		83	30	37	58	225	998	271	199
	3			17	212	159	65	64	46	14	69	113		71	29	83	92	111	168	255	215
	4			4	102	162	53	78	39	37	14	96		27	20	55	182	46	145	51	241
	5			1	22	72	47	39	37	12	22	27		3	6	44	95	43	51	45	50
	6			11	23	20	17	48	16	24	5	24		6	2	21	83	14	66	25	46
	7			5	25	15	2	15	23	14	24	20		3	1	1	27	20	27	51	28
	8			5	15	21	2	5	16	15	14	19			1	4	12	6	31	34	25
	9			3	7	17	7	1	4	4	12	22		1		7	10	2	6	21	34
	10			1	3	14	4	9	1	1	7	8		2	2	2	16		5	13	21
	11				3	11	13	3	4	2	3	7				3	9	1	2	3	7

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Background

The species profiles herein are a selection of 23 of the some of the most important fishery species of northern Australia. Although there are many others that could have been included, the species were selected to be representative of the regions, fishery sectors and taxa, while also being identified as high priority species during consultations with stakeholders. As a companion report to *Part 1: Vulnerability assessment and adaptation options,* the information compiled here for each species provided the necessary baseline information for this project: (i) carry out further species sensitivity data analyses, (ii) conduct the species-based vulnerability assessments, and (iii) identify appropriate adaptation options and barriers. Each species profile covered the following aspects: fisheries, biology, ecology and life cycle, and environmental sensitivity and resilience in a climate change context. This content followed the template set by the similar project conducted in south-eastern Australia (Pecl et al. 2011) thereby ensuring consistency across projects.

Each profile involved comprehensive literature reviews so as to provide the most up-to-date, and therefore relevant, information to inform the major tasks of the project. Firstly, identifying the known sensitivity of each species to key environmental (climate) variables helped us to set up hypotheses for testing for the data analyses conducted for some species, determined the information gaps, and informed the development and scoring for the vulnerability assessments. Documenting the biology, ecology and life history also informed the development of the hypotheses as well as the vulnerability assessments. Information about the fisheries, including their management and operational characteristics, was important also in informing the vulnerability assessments, and particularly in identifying adaptation options for fisheries.

This report should represent a useful and interesting stand-alone resource for any fishing stakeholder group in northern Australia.

Reference:
Pecl GT, Doubleday Z, Ward T, Clarke S, Day J, Dixon C, Frusher S, Gibbs P, Hobday A, Hutchinson N, Jennings S, Jones K, Li X, Spooner D, and Stoklosa R (2011). Risk Assessment of Impacts of Climate Change for Key Marine Species in South Eastern Australia. Part 2: Species profiles. Fisheries and Aquaculture Risk Assessment. Fisheries Research and Development Corporation, Project 2009/070.

INVERTEBRATES

1. Banana prawn, *Penaeus merguiensis*

Authors: Julie Robins and David Vance

Banana prawns belong to the family *Penaeidae*. Over 50 different species of penaid prawns occur in Australian waters, with about 10 species of major economic importance. Despite all belonging to the same family, commercially important prawns in northern Australia have distinct differences in their distribution, habitat preferences, seasonality, recruitment dynamics and migratory abilities. These differences suggest that Australian penaeid prawn species are likely to have different sensitivities and resilience to any changes in environmental conditions resulting from long-term global climate change.

The fishery

Key points:
- The two key fisheries for banana prawns can be divided into the Queensland east coast (river and estuarine beam trawl fishery, and the offshore otter trawl fishery) and the Northern Prawn fishery, which operates from the Gulf of Carpentaria to Cape Londonderry in Western Australia.
- It is estimated that the Northern Prawn Fishery harvests ~90% of the annual banana prawn population.
- In northern regions catch is influenced by SOI and rainfall/riverflow, while temperature is likely to play a more significant role in southern regions.

Queensland east coast

Banana prawns are taken by commercial and recreational fishers on the Queensland east coast. Commercial fisheries include the river and estuarine beam trawl fishery, which harvests regional sub-stocks of juvenile and sub-adult banana prawns, and offshore otter trawls, which harvest schools of sub-adult and adult banana prawns. Offshore schools of banana prawns are generally

associated with major river systems and can be geographically grouped into the following sub-stocks (Tanimoto et al. 2006): Cooktown, Cairns, Tully, Townsville, Mackay, Fitzroy, Gladstone, Burnett, and Moreton. The annual harvest of banana prawns in the otter trawl fishery is variable, ranging from 230 t to 978 t, with an average of ~500 t. The annual harvest in the beam trawl fishery is more stable, ranging from 71 t to 235 t, with an average of 133 t. There is also a relatively small commercial stripe net fishery for banana prawns in the Burnett and Mary River systems. Recreational fishers also harvest bananas prawns via cast netting in estuaries and near shore areas adjacent to major population centres. The recreational harvest varies between years and regionally, but is thought to be in the order of ~100 t (Tanimoto et al. 2006) .

The multi-species Queensland East Coast Otter Trawl Fishery occurs from the border between Queensland and New South Wales northwards to the Torres Strait. This fishery is managed by input controls including limited entry, net and mesh size regulations, individual effort limits, vessel restrictions, and spatial and temporal closures. However, there are no specific input controls for banana prawns. Banana prawns are predominantly harvested during daylight and are different to others species captured within the fishery in that a significant amount of the fishing effort is spent "searching" for aggregations of banana prawns.

Northern Prawn Fishery

Banana prawns are a commercial only harvest in the Northern Prawn Fishery, a commonwealth managed fishery between Cape Londonderry (Western Australia) and the western tip of Cape York (Queensland). Two species of banana prawns are caught in the fishery. The common banana prawn, *Penaeus merguiensis*, is caught throughout much of the fishery while the Indian banana prawn, *P. indicus,* is caught in two locations; in Joseph Bonaparte Gulf and in a small area just north of Melville Island. Banana prawns are harvested over a tightly controlled season beginning in April and which can last between 6 and 14 weeks, depending on the banana prawn abundance and subsequent catch rates. The catch of banana prawns in the Northern Prawn Fishery is variable, with catches around ~5,800 tones for 2008, 2009 and 2010; ~7,100 tonnes in 2011 and ~4,900 tonnes in 2012 (AFMA 2013). The fleet of the Northern Prawn Fishery is thought to harvest around 90% of the annual banana prawn population. Although two species of prawns are caught in the fishery, they both have similar estuarine habitat requirements as juveniles and their responses to climate change will probably be similar.
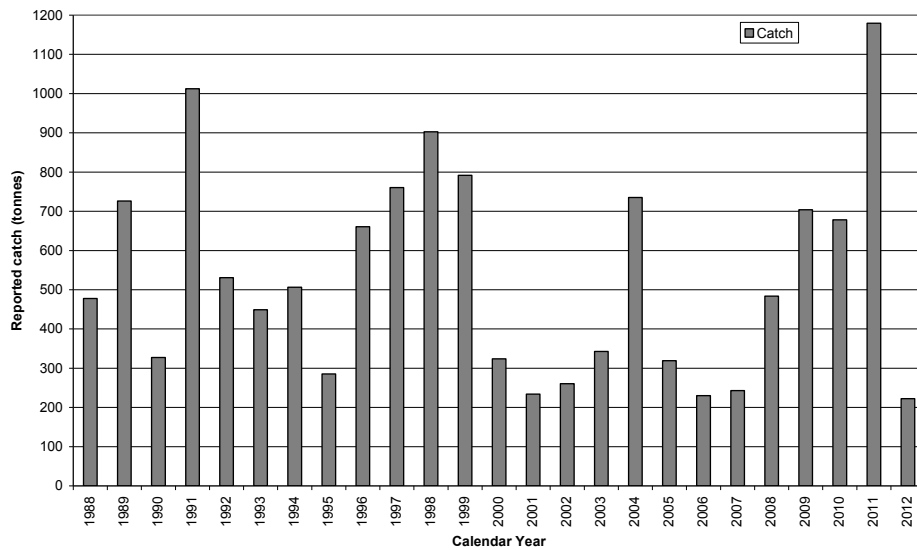
Figure 1.1. Reported catch of banana prawns for Queensland East Coast Otter Trawl Fishery.

Life history

> **Key points:**
> - Banana prawns appear to be able to adapt to local environmental conditions, with Indian banana prawns from the Red Sea adapted to the high salinity conditions.
> - Recruitment is spatially variable, and probably linked to rainfall and riverflow.
> - Sub-stocks exist along the Queensland east coast and the Gulf of Carpentaria with limited exchange between sub-stocks.

Life cycle, age, growth and environmental variation

Banana prawns are an estuarine and coastal species, and as adults are associated with waters up to 20 km from the coast and up to 45 m depth (Grey et al. 1983). Banana prawns have a typical type-2 penaeid life cycle (Dall et al. 1990). Adolescent banana prawns migrate downstream from estuarine habitats to marine waters. Here, they mature and spawn eggs which are demersal for less than a day, before becoming pelagic larvae. Larvae and post-larvae migrate from offshore waters into estuaries using tidal currents, and settle as post-larvae in mangrove-lined estuarine nursery habitats. Juvenile banana prawns remain in the estuary for several months, before migrating out of the estuary to coastal marine waters.

Banana prawns can spawn throughout the year if water temperatures are appropriate. In northern Australia, there are usually peaks of population spawning in autumn, when individuals spawned in the previous spring mature into adults, and also in spring, when water temperatures warm up after winter. Spring-spawned individuals migrate into nursery habitats between November and March and it is likely that they contribute most to the autumn commercial fishery for banana prawns in coastal and offshore waters of both the NPF and the Queensland ECOTF.

Banana prawns are thought to have a one-year life cycle although some older prawns are caught in commercial catches. A complicating factor is the report by Dredge (1985) of the recruitment of juvenile banana prawns (>10mm carapace length, CL) into the Burnett River in southeastern Queensland from December to March as well as May to June. Large banana prawns have been recorded in the estuaries of central and southern Queensland during winter and there is some speculation that this is indicative that banana prawns in this area may have a six-month life-cycle with two generations per year (Dredge 1985).

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

Temperature, as in all crustaceans, affects various aspects of the life cycle of banana prawns. Spawning of adult prawns in the Gulf of Carpentaria occurs over a wide range of water temperatures but seems to be particularly stimulated by increasing temperature during spring time (Crocos and Kerr 1983). They also found that the maximum proportion of females spawning in the north eastern Gulf occurred in January when water temperatures were around 30°C. It is not known what maximum temperature would prevent banana prawns from spawning at all.

Nauplia, protozoeal and mysis stages of *P. merguiensis* spawned from adult prawns caught in Pakistan had the highest survival in 30 to 35 ppt salinity (Nisa and Ahmed 2000), while in India, the best hatching rate of eggs was found at 33ºC and salinity of 35 ppt (Zacharia and Kakati 2004). They also found that survival rates after hatching were higher at 33ºC and salinity of 35 ppt.

There is no evidence of temperature affecting the migration of banana prawn postlarvae into estuarine nursery grounds but low estuarine salinities do seem to prevent the immigration of postlarvae to estuaries in the Gulf of Carpentaria (Staples 1980a; Vance et al. 1998a).

Haywood and Staples (1993) reported that salinity had no detectable effect on growth rates of juvenile banana prawns. However, in a laboratory experiment, juvenile banana prawns were found to have optimal food consumption and production at 20 ppt salinity, while at higher salinities there was a considerable decrease in growth and food consumption (Vinod et al. 1996). In contrast, Saldanha and Achuthankutty (2000) report that growth of juvenile banana prawns increased with salinity (up to 40 ppt). Staples and Heales (1991b) reported that the optimum temperature and salinity for the growth in length of juvenile banana prawns (i.e., shortest intermolt period and largest moult increment) was 31ºC and 30 ppt salinity (resulting in a weekly growth rate of ~1mm/week). However, taking into account survival, the optimum temperature and salinity for the greatest increase in biomass and production were 28ºC and 25 ppt salinity. Staples and Heales (1991b) concluded that deviations from the optimum temperature had a greater effect on productivity than

changes in salinity. Based on their experimental work, Staples and Heales (1991b) predicted that in an estuary, postlarval prawns would grow quickly but suffer high mortality when temperature and salinity were high, but would grow slowly and remain in nursery areas if the salinity of the estuary fell below 20 ppt**.**

One of the most interesting studies was by Kumlu and Jones (1995). They examined the growth and survival of *P. indicus* using postlarvae reared in the laboratory from brood stock that originated in India. They used the same experimental protocol as had been used by Bukhari *et al.* (1994) who studied *P. indicus* bred from adults caught in the Red Sea. Water temperatures were between 29 and 31ºC and salinities tested ranged from 10 to 50 ppt. Kumlu and Jones (1995) found that these hatchery-reared postlarvae of *P. indicus* tolerated a wide range of salinities. For the smaller postlarvae, up to about PL20, the lower salinities produced the best growth and survival. However, from PL20 to PL60, the growth and survival of the Indian postlarvae was highest at salinities of 20 and 30 ppt, whereas for the Red Sea postlarvae, the highest growth and survival was at salinities of 35 ppt and higher with maximum yield at 50 ppt.

It would appear that there are two distinct strains of *P. indicus;* the Indian laboratory postlarvae behaved similarly to wild-caught Indian postlarvae while the Red Sea postlarvae seemed to be adapted to the much higher salinities that occur naturally in the waters of the Red Sea. The results suggest that banana prawns have the capacity to adapt to different environmental conditions. Emigration of juvenile and adolescent banana prawns from estuaries occurs mostly during the wet season and is highest immediately following high rainfall (Staples and Vance 1986; Vance et al. 1998a). This response is probably mediated by a physiological response of the prawns to low salinity (Dall 1981).

Commercial catches of banana prawns in some regions of the Northern Prawn Fishery are highly correlated with rainfall during the previous wet season (Vance *et al*. 1985; Vance *et al*. 2003). This high correlation is a consequence of the increased emigration of prawns from estuaries during periods of high rainfall (see previous paragraph). Vance *et al*. (1985) also noted a negative correlation between wet season temperatures and the annual commercial banana prawn catch, but this should not be interpreted as a direct effect of temperature on prawn catches. It is simply a result of the negative relationship between rainfall and temperature; high rainfall during the wet season results in lower air temperatures.
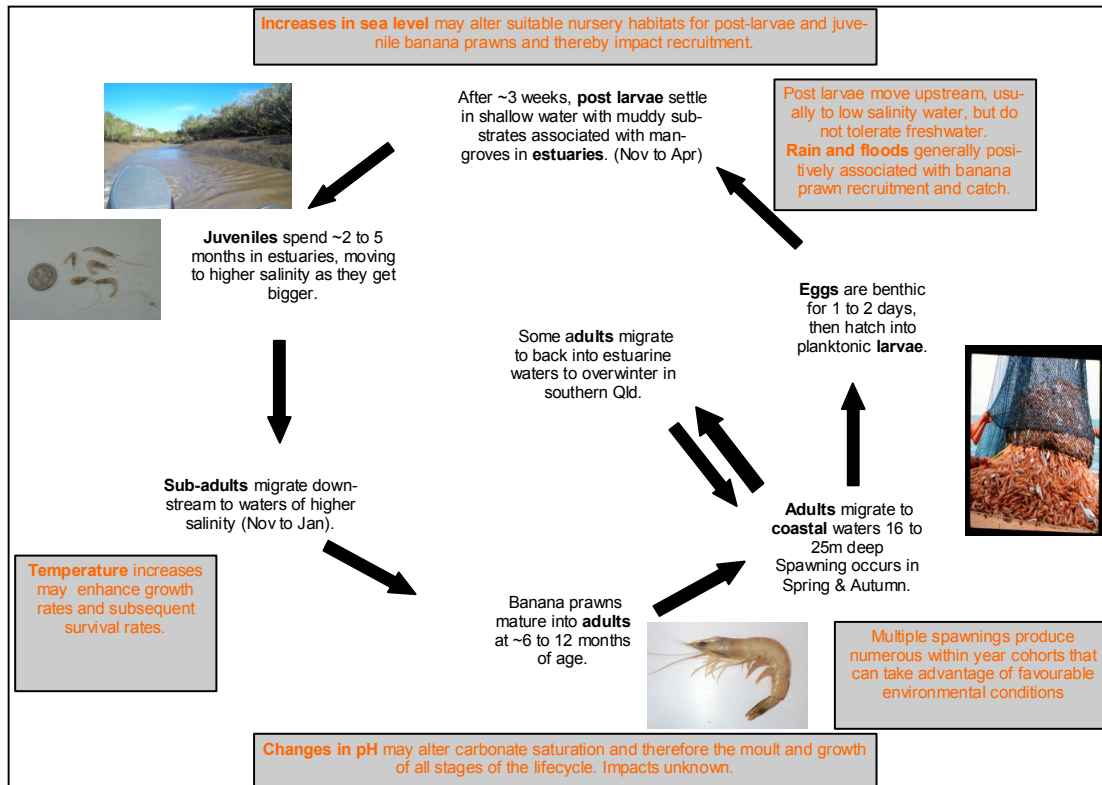
Figure 1.2. Generalised life cycle of the banana prawn. (Images sourced from QDAFF).

Distribution, habitat and environmental preferences

Banana prawns are distributed in tropical and sub-tropical areas from Shark Bay (Western Australia) to Northern New South Wales (Figure 1.3). The benthic post larvae and juveniles are usually associated with estuaries that have mangrove-lined muddy banks and freshwater influences. Banana prawns are very rarely found in locations in estuaries where there are no mangroves. The small prawns move into mangrove forests as the water level rises on high tide and then move out into the open rivers and creeks as the water level falls towards low tide. Vance *et al.* (2002) sampled inside mangrove forests over several years and concluded that, in general, the fringing parts of the mangrove forests were used more by the banana prawns when they were inside the mangroves at high tide.

Juvenile and sub-adult banana prawns gradually migrate downstream as they increase in size and emigrate from the estuaries to coastal waters in summer and autumn at times of high seasonal rainfall and decreases in estuarine water salinity (Staples and Vance 1986; Meager et al. 2003b; Halliday and Robins 2007). Adults are trawled offshore in schools at depths between 16 and 25 m (Tanimoto et al. 2006).
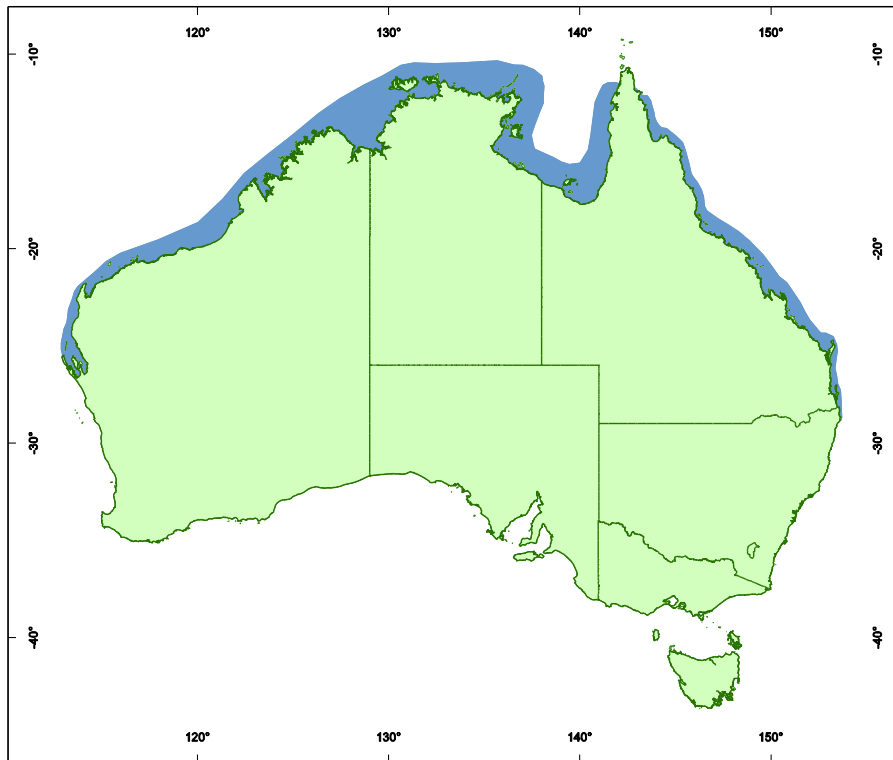
Figure 1.3. Australian distribution of the banana prawn, *P. merguiensis.*

Predators and prey

Banana prawns are an important part of the food chain and are eaten by many species of fish, including some of commercial value. Salini *et al.* (1990) and Salini *et al.* (1998) found that juvenile *P. merguiensis* were important prey of many fish species in two estuaries in the Gulf of Carpentaria. In the northeastern Gulf of Carpentaria, banana prawns were eaten by ten out of the 26 predators that were caught in good numbers in the estuary, including the highly valuable barramundi, *Lates calcarifer*. Salini *et al.* (1998) also found that fish seemed to target banana prawns at times when the prawns were more abundant. Robertson (1988) also found that juvenile *P. merguiensis* were eaten by several fish, including young barramundi. Other studies didn't identify prawns to species but also identified that penaeid prawns were an important part of the diet of some fish in estuaries in the Northern Territory (Davis 1985) and on the Queensland east coast (Russell and Garrett 1983). Adult banana prawns on the commercial fishing grounds of Albatross Bay in the Gulf of Carpentaria were a significant component of fish diets. Brewer et al. (1991) estimated that predators of banana prawns (i.e., fish) consume about 3x as many banana prawns as are harvested by the Northern Prawn Fishery.

Whilst in estuaries, juvenile banana prawns are carnivorous detritivores, consuming a wide range of organisms and organic detritus (Chong and Sasekumar 1981). Gut-content studies report unidentified debris as well as live benthic and pelagic animals such as polychaetes, copepods, amphipods, isopods, mysids, carids, sergestids, foraminifera, molluscs, gastropods, nematodes, insects, diatoms, algae, bacteria, epiphytes (Wassenberg and Hill 1993). Banana prawns feed while inside mangrove forests as well as in the shallows of creeks and rivers when the water levels are

below the mangroves (Logan River – Sue Pillans, personal communication). Newly arrived pelagic post-larvae are carnivorous, feeding mostly on calanoid copepods, while epibenthic post-larvae and juveniles are carnivorous detritivores feeding on detritus, foraminiferans (Rhotallidae), copepods (calanoid and harpacticoid), larval bivalves, diatoms and brachyuran larvae (Chong and Sasekumar 1981). Subadults are detritivorous carnivores feeding on large crustaceans such as *Acetes* and mysids, with lesser amounts of detritus. Adults are detritivorous carnivores feeding on detritus and animals (e.g., large crustaceans *Acetes*, molluscs and fishes) in equal amounts. Plant material consumed by juveniles (in small but consistent amounts) included pieces of mangrove, filamentous algae (*Trichodesmium* and *Microcoleus* spp), and diatoms (*Coscinodiscus*, *Cyclotella*, *Pleurosigma* and *Gyrosigma* app.)

Isotope studies are suggested to give a better indication of the relative importance of dietary items because results indicate a time-integrated, objective measure of carbon assimilated by the organism (Primavera 1996). Several authors have investigated the isotopic signature of banana prawns to identify the relative importance of the various organisms in the nutrition of banana prawns. Newell *et al.* (1995) reported that juvenile banana prawns living in tidal creeks derived nutrition from mangrove sources as well as benthic microalgae, although the greater relative abundance of mangrove detritus in tidal creeks resulted in its greater consumption by juvenile banana prawns. Primavera (1996) reported that δ^{13}C of banana prawns (-18) was closer to plankton and epiphytic algae (-22.6 and −24.3 respectively) than to mangroves (-28.6). She reported a similar finding for δ^{15}N, with banana prawns (<9 mm to 30 mm CL) having a signal (6.9) closer to epiphytic algae (6.0) than to decomposing mangrove leaves (3.8) or plankton (2.3). Primavera (1996) noted that the high δ^{15}N for epiphytic algae may be due to contamination by nematodes and meiofauna present in the samples. Primavera (1996) suggested that the enriched δ^{15}N signal of banana prawns suggests that prawns are two to three levels up the trophic chain from phytoplankton (assuming a 2.4% enrichment per trophic level). Primavera (1996) suggested the use of stable S to improve the understanding of plankton-penaeid shrimp connections. Loneragan *et al*. (1997) found similar results to those of Primavera (1996). Banana prawns had δ^{13}C and δ^{15}N values closer to that of macroalgae/seston. Values of δ^{34}S were between the values of a seagrass (*E. acoroides*) and a mangrove (*C. tagal*). They concluded that juvenile banana prawns were likely to obtain <10% of their nutrition from mangrove detritus.

Recruitment
Banana prawns recruit in multiple cohorts into estuaries between November and May.

Current impacts of climate change
There are no documented impacts of climate change on banana prawns, although there are documented links between harvests and river flow/rainfall.

Sensitivity to change

Key points:
- Banana prawns are highly reliant on mangrove-lined mud banks in estuarine areas for postlarval growth and survival and negative consequences on mangrove distribution due to sea level rise would result in decreased banana prawn abundance.
- Banana prawn growth and survival has been shown to be optimal at 28 °C compromised at temperatures of 35 °C. Increase in SST may impact on banana prawn abundance in far northern areas.

Many species of prawn show relationships between catch and environmental variables (see Dall et al. 1990 for review; Vance et al. 2003; Halliday and Robins 2007) but these relationships are not always consistent over time and space. Prawns also prefer optimum environmental conditions (e.g. salinity, temp) and these optimums vary between prawn species, reflecting their life-cycle preferences as classified by Dall *et al.* (1990). Banana prawns inhabit estuaries and as such, are already exposed to daily and seasonal fluctuations in many of the physical attributes of shallow waters (i.e., <5m), including water temperature, salinity, turbidity, and pH.

Potential impacts of climate change on banana prawns can be divided into two categories; direct impacts on the survival and/or growth of the prawns, and indirect impacts due to changes in habitat or other characteristics of the environment or ecology.

Direct impacts

Increase in water temperature is the factor most likely to have a direct impact on banana prawns survival and growth. Staples and Heales (1991a) found that the optimum condition for increase in biomass of juvenile prawn populations in the laboratory was at a temperature of 28°C and salinity of 25 ppt. They also found that after six weeks at 35°C and at 35 ppt, all prawns kept under those conditions died. Clearly, there is some potential for impact on banana prawn populations if water temperatures increase substantially. Rothlisberg *et al.* (1998) discussed potential impacts of climate change on banana prawns and felt that increased water temperatures of 2 to 3°C would not significantly affect spawning behaviour of the prawns.

We need to be careful when we interpret some published results on the effects of temperature. For example, Vance *et al.* (1985) found a significant negative correlation between wet season temperature and annual commercial prawn catch. However, wet season temperature is negatively correlated with the amount of wet season rainfall. In fact, it is the rainfall that is driving the catch correlation, not temperature. This is confirmed by detailed biological studies that showed that prawn emigration from the estuaries was highly correlated with rainfall events (Staples and Vance 1986; Vance *et al.* 1998b). Variation in rainfall can have significant impacts on the abundance of adult banana prawns in offshore waters. In some regions of the Gulf of Carpentaria, it would be possible to make quite good estimates of how offshore commercial catches would vary if annual rainfall increased or decreased to particular levels (Vance *et al.* 1985; Vance *et al.* 2003).

As well as variation in total rainfall, banana prawn abundances could be susceptible to changes in seasonal patterns of rainfall. The banana prawns life cycle requires medium to high salinities in the

estuarine nursery grounds in spring and early summer so that postlarval prawns can migrate into the estuaries and spend two to three months growing within the estuary. Increased rainfall and lower salinities during the summer wet season and early autumn then stimulates the adolescent prawns to move offshore again. If the length of the wet season increased substantially and if salinities became low during late spring and early summer then post larval prawns might be prevented from reaching the nursery habitat and therefore not survive through to adulthood.

It is important to note that research on banana prawns in the Gulf of Carpentaria has shown strong correlations between rainfall and offshore adult banana prawn catches in some regions but not in others (Vance et al. 1985). However, more detailed biological research in the estuaries in these regions has shown that prawns in the different regions still emigrate from the estuaries in response to rainfall/low salinity (Vance et al. 1998a). The lack of correlation between rainfall and offshore catch in some regions is not because the prawns behave differently in different regions but because of other factors such as different levels of rainfall in different regions and lower variability in rainfall between years in some regions or difficulties in estimating actual abundances of prawns.

Indirect impacts

The most significant potential indirect impact on banana prawns would be an increase in Mean Sea Level (MSL), which might impact the primary estuarine nursery habitat of juvenile prawns. Juvenile banana prawns are very sensitive to the availability of mangrove-lined mud banks and are only found associated with mangrove-lined mud banks. If MSL increased so dramatically that mangroves disappeared from some areas, then banana prawn abundance would also decrease.

Resilience to change

> **Key points:**
> - Banana prawns are likely to be resilient to increases in temperature projected for northern Australia over at least the medium-term (~50 years).
> - Populations of banana prawns are likely to perpetuate in the face of reduced riverflow, however, future fishery harvest levels of banana prawn may decrease in some of these regions.

Temperature

From laboratory experiments, juvenile banana prawns are unlikely to survive extended periods living in water temperatures of 35°C or greater (Staples and Heales 1991b). During the 1970's to the 1990's, water temperatures at Karumba and at Weipa in the northern Gulf of Carpentaria reached just over 30°C at times (D. Vance personal observation). If water temperatures in northern Australia consistently reach 35°C then it would appear that banana prawn abundances would almost certainly decrease in some areas. However, we must use some caution in extrapolating the results of Staples and Heales (1991b) to banana prawns in far northern Australia. Their laboratory experiments used post larval and juvenile prawns caught from Emu Park, near Rockhampton, central Queensland east coast. It is possible that prawns that have been born and bred in warmer northern waters might survive water hotter than 35°C. Research on banana prawns in other parts of the world has certainly shown that prawns from different regions have different levels of growth and survival when subjected to the same conditions of temperature and salinity (Kumlu and Jones 1995).

If changes in water temperature occur gradually, as we would expect, it is quite likely that prawn populations may be able to adapt to higher temperatures resulting in less change than we would expect based on the results of Staples and Heales (1991). Although increased water temperatures might cause some stress for banana prawn populations in the far north of Australia, an increase in water temperature at the southern end of the range of banana prawns in Australia would almost certainly mean that populations of banana prawns would appear further south than they currently exist. In southeastern Queensland, near the southern end of the range of banana prawns on the east coast of Australia, the seasonal variation in water temperature is higher than in tropical waters, and warm temperatures were actually associated with higher juvenile prawn catches in the Logan River (Meager et al. 2003a; Courtney et al. 2011). Therefore, although the pattern of distribution of banana prawns in Australia might change to some extent in response to temperature change, the overall abundance of prawns may not change much.

Rainfall

As noted above, substantial, decreases in rainfall will probably lead to decreased abundances of adult banana prawns in offshore waters. In some regions of northern Australia we have seen large fluctuations in annual commercial banana prawn catches in response to changes in annual summer rainfall. However, even with very low levels of rainfall and when offshore catches have dropped to near zero, enough adults have survived to reproduce and even produce large populations of banana prawns within one year when rainfall conditions have improved (Vance et al. 2003). Therefore, we believe that although local populations of banana prawns may decrease, the species is sufficiently resilient that populations will survive. If wet season rainfall levels increased then banana prawn abundances would probably increase.

Mean Sea Level and effects on mangrove habitat

It is important to note that not all parts of the mangrove forests are equally important for banana prawns. Vance et al. (2002) found that at high tide, juvenile banana prawns were more frequently caught at the fringes of the mangrove forests than deep inside the forests. Loneragan et al. (2005) found that, in Malaysia, the abundance of juvenile banana prawns caught near narrow fringing mangrove forests, 5 to 10 m wide, was the same as the abundance near wide mangrove forests.

It is likely that, if mean seal level (MSL) does increase, the increase will be gradual. In some areas the boundaries of the mangrove forests may simply shift further back from present river or creek edges. In other areas, the mangrove forests may simply continue to trap sediments as the MSL increases and the substrate level may also increase such that there is very little actual change in the pattern of mangroves. Unless the fringing mangroves are destroyed completely by climate change then we believe that changes in MSL will not impact banana prawn abundance or distribution. However, in areas where there are barriers to mangroves moving landwards with sea level rise (eg. coastal development) there will potentially be localised reductions in banana prawn abundance.

Other authors have come to slightly different conclusions on the potential impacts of MSL rise. Morison and Pears (2012) completed an expert based vulnerability assessment of the Queensland East Coast Otter Trawl Fishery and concluded that banana prawns in Queensland had a high level of ecological vulnerability to sea level rise and changed rainfall patterns, which could result in both positive and negative effects. Morison and Pears (2012) also found that banana prawns had a

medium level of ecological vulnerability to higher sea surface temperatures, with effects on the growth and survival of juveniles.

Rothlisberg *et al*. (1988) assessed the possible impact of the greenhouse effect on commercial prawns in the Gulf of Carpentaria and concluded that, if the greenhouse effect leads to higher sea levels, higher rainfall and increased cyclone activity, we could expect an increase in banana prawn catches.

Other

> **Key points:**
> * Water resource extraction and management, particularly on the Queensland east coast, is a potentially significant additional stressor of banana prawns in estuarine ecosystems.

Ecosystem level interactions
Banana prawns are a key food source for many estuarine-dependent and inshore coastal species. Changes in the distribution and abundance of banana prawns may potentially effect the distribution and abundance of their predators.

Additional (multiple) stressors
Banana prawns are an estuarine-dependent species whose production is linked to river flows. Management of water resources for human use has the potential to exacerbate climate stressors, particularly under scenarios with reduced rainfall as human demand for water resources often takes precedent over ecosystem needs.

Critical data gaps and level of uncertainty
One of the critical data gaps is how banana prawns respond to altered pH at various stages within their life cycle.

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2. Eastern king prawn, *Melicertus (Penaeus) plebejus*

Authors: Julie Robins and Tony Courtney

Eastern king prawn, *Melicertus plebejus*. Photo: T.C. Courtney.

Eastern king prawns are predominately found on the east coast of Australia and are a single stock that is shared between Queensland and New South Wales. Although there are arrangements for collaborative stock assessment, the fishery is managed separately under each the state's jurisdiction.

The fishery

> **Key points:**
> - Eastern king prawns are a shared resource between Queensland and New South Wales, with Queensland producing about 80% of landings in the last five years.
> - In Queensland the catch is commercial only.

Eastern king prawns are fished only by commercial trawlers in Queensland where they are taken as juveniles and adults in offshore otter trawls. Eastern king prawns form part of the multi-species East Coast Otter Trawl Fishery, which occurs from the border between Queensland and New South Wales northwards to the Torres Strait. Eastern king prawns are harvested across their distribution in Queensland, which is predominately south of 22°S. The eastern king prawn catch is worth around $32 million, based on an annual average catch of 2000 t and a price $16/kg received by fishers. The economic viability and profitability of the Queensland East Coast Otter Trawl Fishery is dependent on several factors, one of which is fuel price, and varies between sectors as there is a price differential for different species of prawn.

Management
The Queensland East Coast Otter Trawl Fishery is managed by input controls including: limited entry, net and mesh size regulation, individual effort limits, vessel restrictions, and spatial and temporal closures. However, there are no specific input controls for eastern king prawns or any other targeted

species in the fishery. The fishery is managed under the Fisheries (East Coast Trawl) Management Plan 1999, with ten different endorsements for different aspects of the fishery. The fishery is also constrained by the zoning plans of the Great Barrier Reef Marine Park, Great Sandy Strait Marine Park and Moreton Bay Marine Park.

Operational characteristics

Trawl nets are used to capture prawns, scallop and several other allowed species in the fishery. Different net configurations are used in different sectors of the fishery, depending on the target species and the type of grounds fished. There are restrictions on the allowable combined head rope length and trawl net mesh sizes, with a limit of 24 fathoms head rope length in shallow-water and 50 fathoms head rope length in deep-water (O'Neill et al. 2003). Vessels targeting eastern king prawns in offshore waters generally use 'triple rig' gear i.e., three nets linked together in between two outer boards, and two inner sleds. This is a very stable configuration especially for trawling in waters up to about 300m deep and in strong currents. Trawling for eastern king prawn occurs at night, when this species emerges to feed from being buried in the sediment.

Vessels in the fishery have a maximum allowable size of 20 m. Around 350 trawl vessels participated in the fishery in 2008 (Anon. 2010), with not all boats participating in all sectors. Vessels vary in their characteristics that affect fishing power, i.e., technologies such as engine power, propeller nozzle, global positioning systems, plotters, and type of otter board. See O'Neill *et al.* (2003) for detailed descriptions of the fleet.

Catch and effort of eastern king prawns is recorded by fishers as part of a compulsory daily logbook program managed by Fisheries Queensland, and commenced in 1988. Eastern king prawns are also part of the Long-Term Monitoring Program by Fisheries Queensland. Fishery-independent surveys of eastern king prawn pre-recruit abundance have been conducted annually since 2006. Details of the survey can be found in Courtney *et al.* (2011).
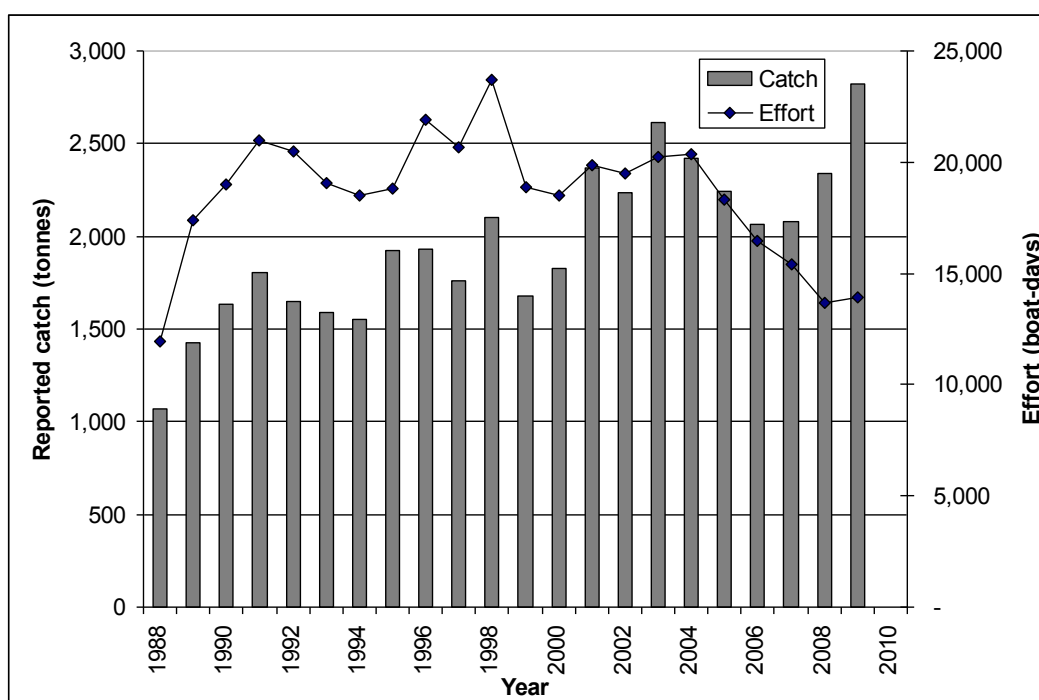
Figure 2.1. Eastern king prawn commercial catch (tonnes) and effort (boat days) by year for Queensland East Coast Otter Trawl fishery.

Life history

> **Key points:**
> - Recruits are probably sourced from a wide number of spawning locations along the Australian east coast.
> - Although spawning occurs at multiple sites and times, recruitment of eastern king prawns to the fishery in Moreton Bay occurs between October and December.
> - Eastern king prawns have an offshore preference as adults with early life history stages predominantly closer inshore.

Life cycle, age and growth

Eastern king prawns have a typical penaeid prawn life cycle of planktonic larvae, pelagic and then benthic postlarvae, juveniles and then adults. Adults spawn in offshore waters >90 m between January and August and spawning is thought to occur near the edge of the continental shelf (Ruello 1975; Courtney et al. 1995b; Courtney 1997c; Montgomery et al. 2007). The peak spawning season is between May and July. Eastern king prawns show extensive spatial population egg production (based on ovary weight, ovary histology, abundance of spawners) from NSW to central Queensland (Montgomery et al. 2007). There are likely to be multiple spawning grounds for eastern king prawn along the coastline of Queensland and New South Wales, with the Swains Reef Complex thought to be a major spawning ground (Courtney 1997a). Multiple spawning grounds provide this species with flexibility for inter-year variation in "effective" spawning areas i.e., areas providing recruits in any one year.

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The distribution of pelagic eastern king prawn larvae into estuarine embayments (Ruello 1975; Courtney et al. 1995a; Rothlisberg et al. 1995) and the recruitment of eastern king prawn into the trawl fishery are almost certainly influenced by the south flowing East Australian Current and its eddies, as well as the predominately onshore winds between September and November in Queensland (Courtney et al. 1996). The benthic post-larvae settle on to bare substrates and seagrass (Young and Carpenter 1977). The post-larvae of eastern king prawn have an aversion to areas with freshwater influence, with fewer individuals settling in these areas and those that do only remaining at sites near river mouths for brief periods (Coles and Greenwood 1983).

Eastern king prawns spend about three months in estuarine embayments (Courtney 1997b) before migrating to ocean habitats (Lucas 1974). They mature into adults at 35 to 40 mm carapace length (CL) (Courtney et al. 1996) when they are between six to 12 months of age. Eastern king prawns can undertake extensive northerly migrations to as far north as the Swains Reef complex (Ruello 1975). Some individuals migrate more than 1300 km (Montgomery 1981).

Although eastern king prawns use embayments as nursery habitats, it is possible for their life cycle to be completed in offshore waters (Montgomery 1990), although this probably requires some type of shallow nursery habitat. Eastern king prawns have a lifespan of up to three years (Ruello 1975), but the majority of the population is thought to live for less than one year. Eastern king prawns have relatively fast growth rates (Glaister et al. 1987).

Distribution, habitat and environmental preferences
Eastern king prawns are found only on Australia's east coast and are distributed from Mackay to Tasmania in depths of 1 - 300 m (Courtney unpublished data). Adults have an offshore distribution, whilst juveniles are distributed throughout ocean-influenced estuaries between southern NSW and Bustard Head in central Queensland (Williams 2002), Queensland (Figure 2.3).

Courtney *et al*. (2011) investigated the influence of abiotic parameters on the standardised catch rates of eastern king prawns within Moreton Bay, Queensland. They reported that average daily flow of the Brisbane River, from one month, two months and three months prior to the catch date significantly affected the daily catch rates of eastern king prawns as did the mean daily maximum air temperature at Cape Moreton for the proceeding 60 days and the proceeding 120 to 180 days. Analyses indicated that abiotic parameters had significant but small effects on the catch rates of eastern king prawns. Flow one and two months prior and temperature in the preceding 60 days all had a positive influence on eastern king prawn catch. Flow three months prior to the catch had a negative influence on eastern king prawn catch. Temperature 121 to 180 days preceding the catch was considered to have the largest significant (negative) effect on the daily catch of eastern king prawns in Moreton Bay. This translated to warmer winter (i.e. April to August) temperatures resulting in lower than expected catch rates of eastern king prawns in October to December.
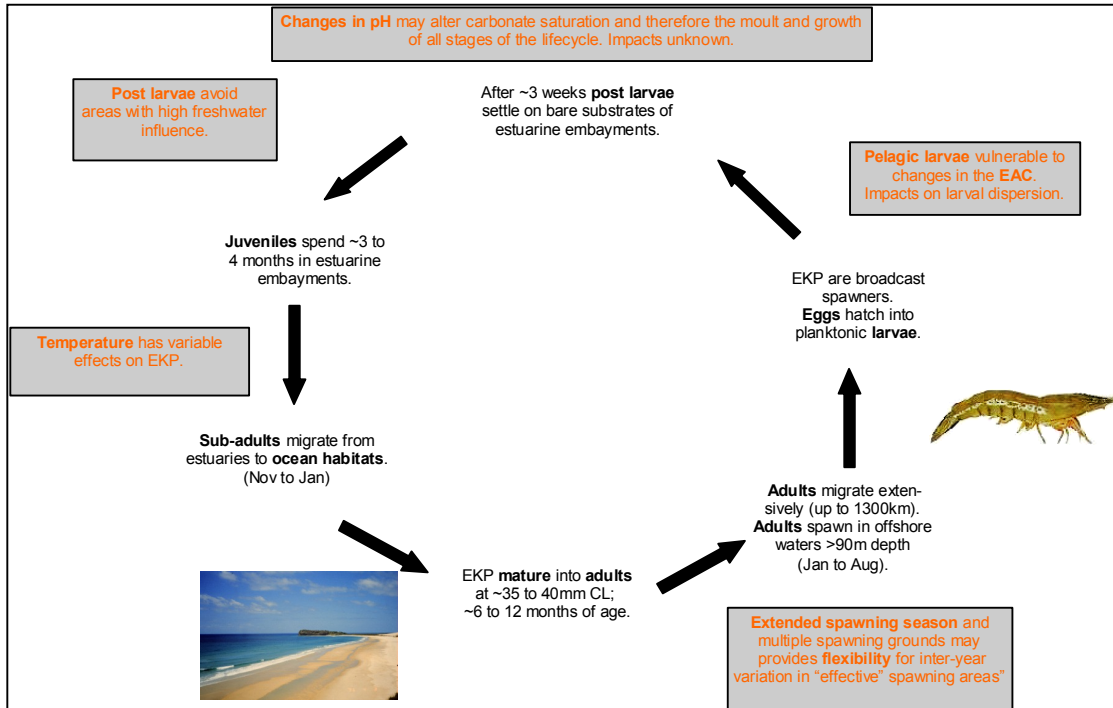
Figure 2.2. Generalised life cycle of eastern king prawn. (Images sourced from QDAFF).
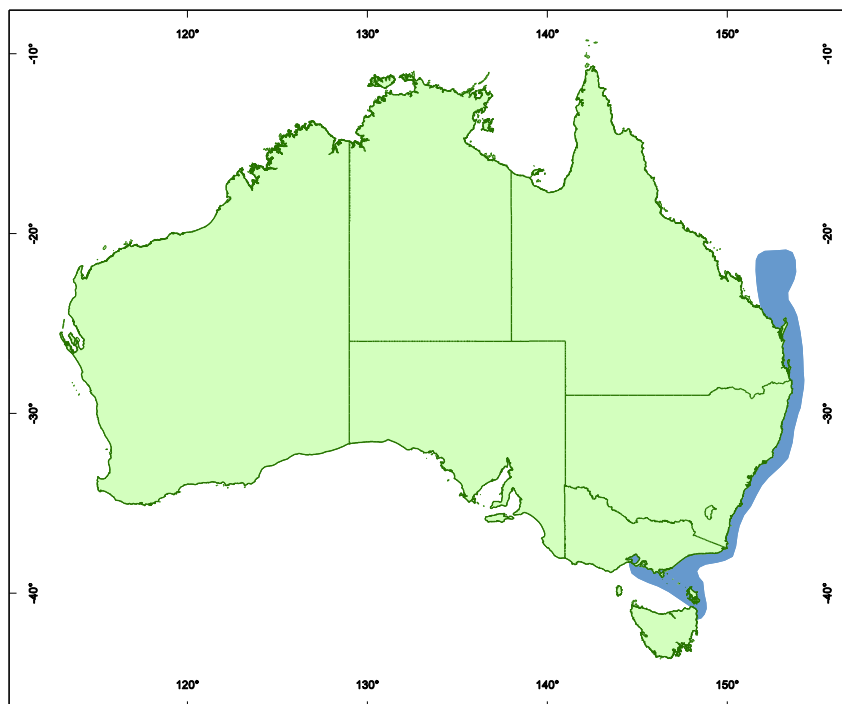
Figure 2.3. Australian distribution of the eastern king prawn.

Predators and prey

Eastern king prawns are opportunistic omnivores, whose diet includes small crustaceans, polychaetes, bivalves and protozoans (Kailola et al. 1993). Predators of eastern king prawns include marine fish.

Recruitment

Spawning of eastern king prawns occurs over many months throughout their distribution, but especially so in the northern part of their distribution, and with greater egg production at lower latitudes (Montgomery et al. 2007). However, it is likely that most egg production is wasted or ineffective due to high mortality rates of eggs, larvae and postlarvae. May to August is a key period for effective egg production and results in the recruitment of juveniles to the Moreton Bay part of the Queensland fishery that is succinct and consistent i.e., October to December (Courtney et al. 1995a).

Courtney *et al*. (1995a) also reported that catch rates of eastern king prawn recruits (< 15 mm CL) and post-recruits (> 15 mm CL) peaked with salinity, and were significantly (P< 0.01, n = 216) and positively correlated with salinity over the 2-year sampling period. This may a coincidence in timing between recruitment and the lack of rainfall (and thus high salinities). When times of peak recruitment were analysed (i.e., September to December), bottom salinity and temperature had no significant effect on the abundance of recruits or post-recruits (Courtney *et al.* 1995a).

Ruello (1975) speculated a strong causal relationship between the East Australian Current (EAC) and the distribution and abundance of eastern king prawns. Rothlisberg *et al.* (1995) challenged this speculation, instead suggesting that recruitment to inshore nursery habitats was derived from localised spawning. The role and importance of the EAC in the population dynamics, movement and distribution of eastern king prawns is still poorly understood and not quantified.

Current impacts of climate change

> **Key points:**
> - The distribution of eastern king prawns is likely to be affected by changes in the East Australian Current, water temperature, sea level rise, changed rainfall and ocean acidification.
> - Exposure differs for estuarine post-larvae and juveniles to that of oceanic sub-adults and adults.

There are no documented impacts of climate change on eastern king prawns. However, Montgomery (1990) speculated that a more consistent southward flow of the EAC below 32°S may lead to less variable recruitment to southern estuaries and a range extension. Montgomery (1990) also speculated that an increase in water temperature of two to four degrees Celsius could trigger: (i) earlier emigration from inshore nursery areas (assuming that emigration is related to water temperature) which would lead to increased aggregation of individuals and to increased catch rates; (ii) earlier spawning (assuming spawning is related to water temperature and ignoring any role day-length may have in determining spawning seasons); and (iii) spawning occurring at more southerly latitudes, as overall reproductive potential is currently greatest at lower latitudes (i.e., Ballina north) in autumn (Montgomery et al. 2007).

Courtney *et al.* (2011) reported a significant negative effect of winter temperatures on the daily catch rates of eastern king prawns in the following October to December in Moreton Bay. They suggested that increasing temperatures associated with climate change may result in a decline in the abundance and/or distribution of eastern king prawns in the Moreton Bay/Southern Queensland region.

A sea level rise of 1.2 to 1.4 m was speculated to potentially increase habitats for juvenile eastern king prawns. Montgomery (1990) speculated that increases in wave action as a consequence of increased wind speed south of 36°S may increase the accessibility (and therefore habitat of eastern king prawn) of intermittently open estuaries. Increases in summer rainfall, and subsequent salinity changes and flooding frequency was speculated to not affect survival of juvenile and adult phases, but may stimulate earlier emigration (Montgomery 1990). Siltation changes that affect the substratum of eastern king prawn habitats have the potential to affect the distribution and possibly the abundance of this species.

Eastern king prawns inhabit oceanic waters along the east coast of Australia where the East Australian Current (EAC) is likely to influence their spawning, recruitment and distribution. Changes to the EAC will alter the distribution and recruitment of eastern king prawns.

Ocean acidification due to carbon dioxide dissolving into the ocean may possibly lower the pH of seawater i.e. from a current mean pH of 8.1 to 7.6 in 2100 (Bechmann et al. 2011) and alter carbonate saturation. This may have a potential impact on the moult and replacement of their exoskeleton and therefore affect growth rates. Eastern king prawns exhibited a positive calcification response pattern to elevated levels of atmospheric carbon dioxide (pCO$_2$) under experimental conditions (Ries et al. 2009). How this translates to responses of the population in the wild remains unknown.

Sensitivity to change

Key points:
- Eastern king prawns have a life cycle that is timed to exploit inshore areas as juveniles when these areas are most stable, i.e., between June and October prior to the onset of major rainfall and flood events.

Gibbs (2011) suggested that eastern king prawns were most sensitive to change during their estuarine phases of their life cycle. Montgomery (1990) argued that the period of highest eastern king prawn post-larval abundance is at a time when summer rainfall would be expected to have the least effect upon salinity and therefore the survival of larvae. This is also true of eastern king prawns in Queensland, who utilise estuarine inshore areas between June and October, which is prior to the onset of major rainfall and flood events.

Resilience to change

Key points:
- There are conflicting opinions about the resilience of eastern king prawn to climate change.

Gibbs (2011) concluded that eastern king prawns in New South Wales were resilient to climate change. However, Morison and Pears (2012) completed an expert based vulnerability assessment of the Queensland East Coast Otter Trawl Fishery and concluded that eastern king prawns in Queensland had a high level of ecological vulnerability to altered ocean circulation and a medium level of ecological vulnerability to higher sea surface temperature, increased tropical storm intensity and flooding, and climate variability driven by El Nino Southern Oscillation.

Other

Key points:
- The effect of changes in the East Australian Current and associated eddies along with increasing acidification on banana prawn populations is poorly understood.

Ecosystem level interactions
Eastern king prawns utilise sandy oceanic habitats that are possibly at less risk of change than habitats associated with other penaeids species (e.g. seagrasss or mangroves).

Additional (multiple) stressors
The eastern king prawn stock is a shared resource between Queensland and New South Wales. Changes in fishery dynamics in Queensland are likely to impact upon eastern king prawn resources in New South Wales.

Critical data gaps and level of uncertainty
The role and importance of the East Australian Current in the population dynamics, movement and distribution of eastern king prawns is still poorly understood and not quantified. It is uncertain if changes in the East Australian Current will result in changes to the thermocline, eddies and other physical oceanographic features along the east coast of Australia and how these may affect eastern king prawns and their fishery.

It is uncertain as to whether changes in ocean pH will have impacts on the calcification and moulting of prawns. There is conflicting speculation as to whether prawns will (Raven et al. 2005) or won't (Cooley and Doney 2009) be affected. There is also the possibility that as prawns inhabit estuaries where pH is probably variable, they may be able to tolerate changes in ocean pH.

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3. Red spot king prawn, *Penaeus longistylus*

Authors: Julie Robins

There are over 50 species of penaeid prawn in Australian waters, of which 10 are of major commercial importance. The red spot king prawn, *Melicertus (Penaeus) longistylus,* has a distinctive (often circular) patch of red on its third abdominal segment. The red spot king prawn is distributed throughout the Indo-West Pacific and South China Sea to Malaysia. It often occurs with the blue legged king prawn *M. latisulcatus*. Both species have similar life histories and distributions and are harvested by trawl fisheries throughout northern Australia.

The fishery

> **Key points:**
> - Red spot king prawns are the main target of a regionally important fishery in central to north Queensland that trawls inter-reef habitats.
> - They are a specific target sector in the Queensland east coast otter trawl fishery and are a minor species in the Torres Strait and Northern Prawn Fisheries.

Queensland east coast

The red spot king prawn forms part of the multi-species East Coast Otter Trawl Fishery, which occurs from the border between Queensland and New South Wales, northwards to the Torres Strait. This fishery is managed by input controls including: limited entry, net and mesh size regulation, individual effort limits, vessel restrictions, spatial and temporal closures. However, there are no specific input controls for red spot king prawns prawns. Although red spot king prawns and blue leg king prawns can be caught along the length of the Queensland east coast, the majority of the harvest is taken between 18°S and 21°S (i.e., Bowen to Lucinda) within the lagoon and inter-reef areas of the Great Barrier Reef in water depths of 40 to 60 m (Kailola et al. 1993; Williams 2002). King prawn landings north of ~21ºS are predominantly red spot kings (70%). Separation of king prawns into the various species (i.e., eastern king, red spot and blue-legged) in the compulsory commercial logbook of the Queensland east coast otter trawl fishery has only occurred since 2003

(http://www.dpi.qld.gov.au/28_18391.htm accessed November 2011). Fishing effort for northern king prawn has declined since 1996, and catch rates have increased. The current catch rate is in the upper range of historical catch rates (http://www.dpi.qld.gov.au/28_18391.htm accessed November 2011).

Torres Strait Prawn Fishery
Red spot king prawns are also harvested by the Torres Strait Prawn Fishery, although this species comprises a minor component of the overall prawn harvest i.e., ~7% (Flood et al. 2010). Recruitment of red spot king prawns to the Torres Strait fishery peaks in December (Somers et al. 1987).

Northern Prawn Fishery
Mostly associated with reefal areas throughout northern Australia (Grey et al. 1983), red spot king prawns are rarely caught in the Northern Prawn Fishery (Barwick 2010).
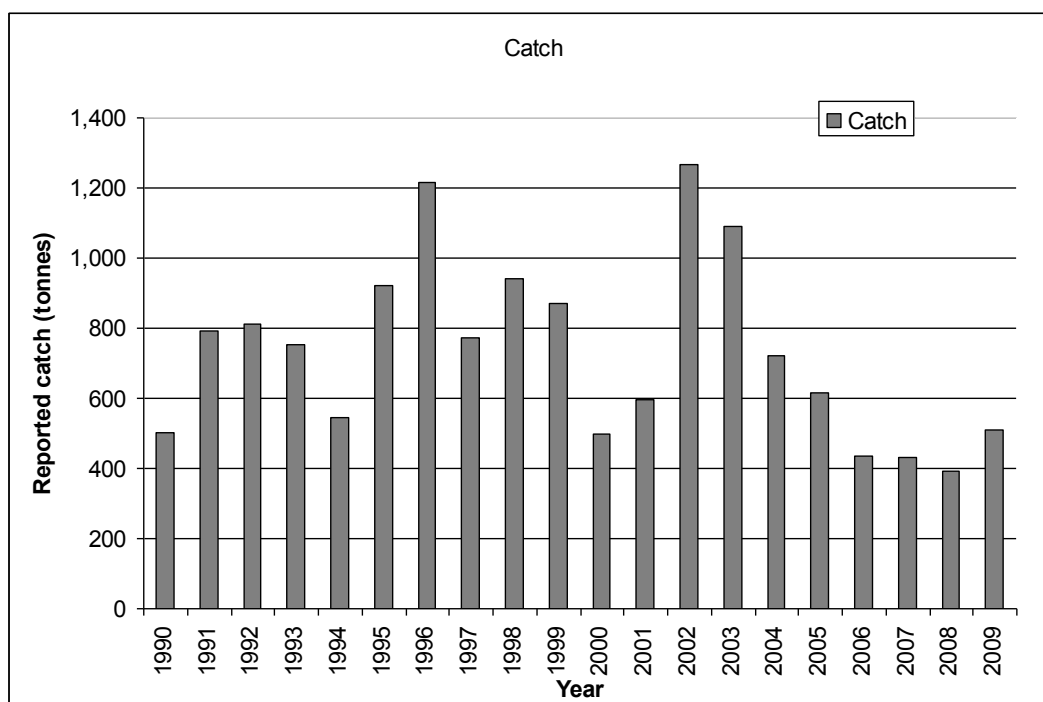
Figure 3.1. Red spot king prawn commercial catch (tonnes) in the northern region of the Queensland East Coast Otter Trawl Fishery (data sourced from Fisheries Queensland).

Life history

Key points:
- Only limited information is known about the biology and environmental requirements of red spot king prawns.
- Differ from other key prawn species in that their life cycle is completely offshore and associated with reef lagoon areas (juveniles) and inter-reef areas (adults).
- Have an extended spawning season (May to October) throughout their northern Australian range.

Life cycle, age and growth

The red spot king prawn has a type-3 penaeid prawn life cycle (Dall et al. 1990), but are atypical in that they are not associated with estuarine or coastal environments. Adults are sedentary and live in inter-reef areas and the lagoon of the Great Barrier Reef on coralline sandy sediments up to 60 m deep. Red spot king prawns have an extended spawning season (May to October), with peak spawning thought to occur between July and August (Courtney and Dredge 1988). Courtney and Dredge (1988) also suggested that there is little geographic variation in spawning periodicity between populations of red spot king prawns that occur in the Gulf of Carpentaria and the central Queensland east coast.

Larvae are pelagic and benthic post-larvae settle between September and May in shallow coralline sandy sediments, often associated with lagoons of coral reefs. Juveniles spend four to six months on reef-tops before emigrating to inter-reef areas. Lunar period does not affect the availability of red spot king prawns to trawl nets unlike other prawn species (Williams 2002).

Like other penaeid species in northern Australia, red spot king prawns are thought to have high natural mortality rates, such that most prawns live for less than two years and a life cycle that is completed within 12 months. Dredge (1990) reported that red spot king prawns have a slower growth rate than eastern king prawns, although this could be a reflection of the timing (i.e., winter) of the study. Growth parameters were also found to vary between males and females and between years.

Distribution, habitat and environmental preferences

Red spot king prawns are distributed in the tropics (Figure 3.3) and use the coralline sand sediment of coral reef lagoons as nursery areas for benthic post-larvae and juveniles. Juveniles migrate from coral reef tops to inter-reef areas in late summer to autumn. Recorded bottom sea-water temperatures during research trawling for red spot king prawns ranged from 23.6°C in September 1985 to 28.5°C in March 1986 (Courtney and Dredge 1988).

Predators and prey

No specific studies on the predators and prey of red spot king prawns are available. However, it would be expected that predators of red spot king prawns would include carnivorous reef fish and that red spot king prawns would be omnivorous eating a variety of bivalves, gastropods, ophiuroids, crustaceans and polychaetes (Wassenberg and Hill 1987).

Recruitment

Recruitment of benthic post-larvae to the lagoons of coral reefs within the Great Barrier Reef occurs between September and May. Juvenile red spot king prawns remain on reefs tops for four to six months, exposing them to the impacts of the Queensland cyclone season i.e., January to April. Regional fishers have expressed concern that the recruitment of red spot kings can be negatively impacted by cyclones (Morison and Pears 2012) but a general dearth of research on this species makes speculation difficult. As red spot king prawns inhabit coral reefs, it may be interesting to investigate whether localised coral bleaching has any impact on localised recruitment and subsequent catch rates of this species.
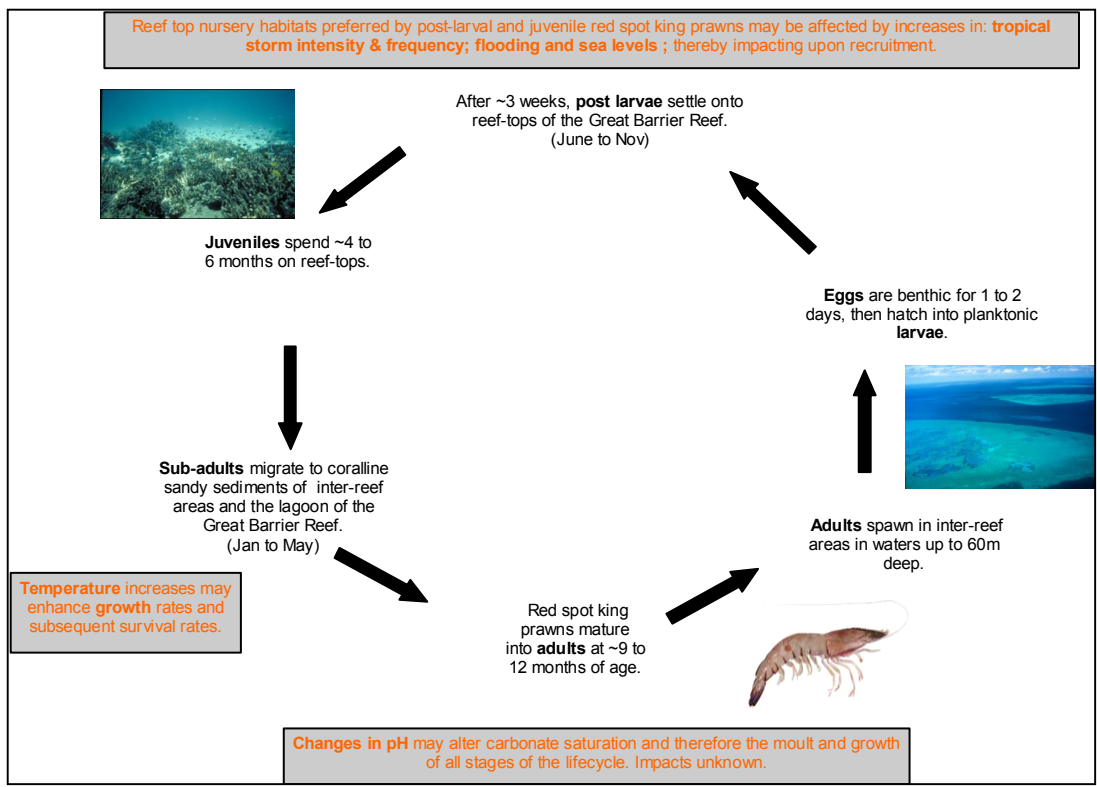
Figure 3.2. Generalised life cycle of the red spot king prawn. (Reef images courtesy of GBRMPA).
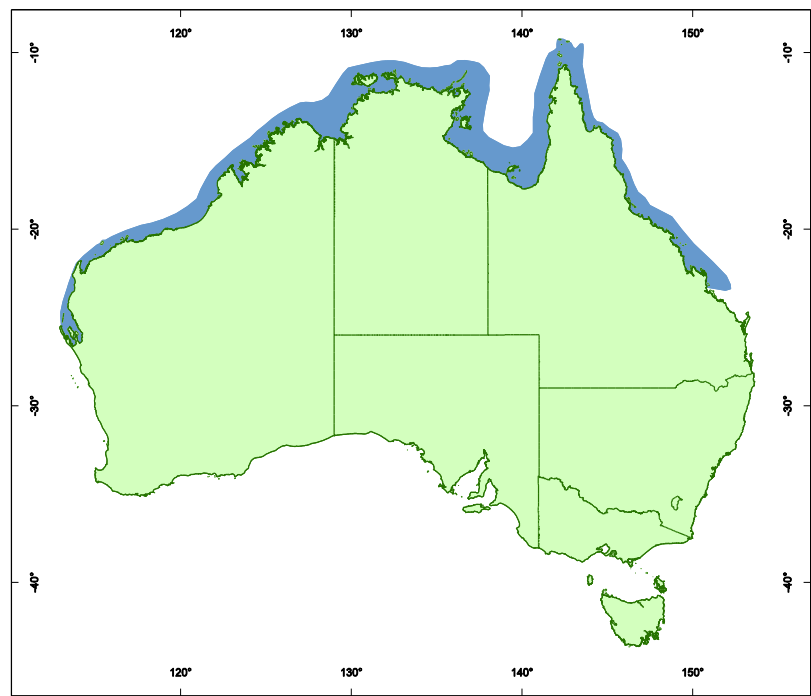
Figure 3.3. Australian distribution of red-spot king prawn.

Current impacts of climate change

There are no documented impacts of climate change on red spot king prawns.

Sensitivity to change

> **Key points:**
> * Potentially sensitive due to specific habitat requirements.
> * Anecdotal reports suggest recruitment of red spot king prawns may be affected by cyclones.
> * As SST rises, there is potential for the distribution of red-spot kings to move southwards, should habitats permit.

Red spot king prawns have specialised habitat requirements as juveniles, potentially making them sensitive to change. An extended spawning season may allow this species to take advantage or cope with changes in some environmental cues such as water temperature.

Resilience to change

> **Key points:**
> * Extended spawning season may allow some flexibility in recruitment.

Morison and Pears (2012) completed an expert based vulnerability assessment of the Queensland East Coast Otter Trawl Fishery and concluded that red spot king prawns in Queensland had a medium level of ecological vulnerability to increased tropical storm activity and flooding which could critically impact on the juvenile reef top habitats. Red spot king prawns also had a medium level of ecological vulnerability to higher sea surface temperature, sea level rise, and ocean acidification.

Other

> **Key points:**
> * Red spot king prawns are the least studied of all the commercial prawn species in Australia.
> * Red spot king prawns are known to have an association with coral reef habitats which are predicted to be degraded under future climate change scenarios.

Ecosystem level interactions

In Queensland, red spot king prawns are associated with coralline sandy habitats, often associated with coral reefs. Degradation of reefs under climate change may impact on habitat quality and or food supply of red spot king prawns.

Additional (multiple) stressors

No specific additional stressors are known.

Critical data gaps and level of uncertainty

Red spot king prawns are the least researched of the main commercial penaeid species of northern Australia. Critical aspects of their biology and possible environmental drivers are unknown.

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4. Brown tiger prawn, *Penaeus esculentus*, and grooved tiger prawn, *P. semisulcatus*

Authors: Julie Robins and Clive Turnbull

Brown tiger prawn, *Penaeus esculentus.* Image from QDAFF.

Two species of tiger prawn are harvested in northern Australia: *Penaeus esculentus,* the brown tiger prawn, and *Penaeus semisulcatus*, the grooved or green tiger prawn. Tiger prawns have similar life histories with both species using seagrass or algal beds as juveniles, but have slightly different distributions as adults. Brown tiger prawns prefer coarse sediments, while the grooved tiger prawn prefers sandy or muddy sediments (Grey et al. 1983; Kailola et al. 1993). Both are harvested by trawl fisheries in northern Australia.

The fishery

> **Key points:**
> - Tiger prawns are a very important component of trawl fisheries in northern Australia with high economic value.

Queensland east coast

Brown and grooved tiger prawns form part of the multi-species East Coast Otter Trawl Fishery, which occurs from the border between Queensland and New South Wales northwards to the Torres Strait. The fishery is managed by input controls including: limited entry, net and mesh size regulation, individual effort limits, vessel restrictions, and spatial and temporal closures. However, there are no input controls for specific to tiger prawns. Although tiger prawns are caught along the length of the Queensland east coast, the majority of the catch is taken north of 21°S in inshore coastal waters (i.e., <30 nm offshore).

Torres Strait Prawn Fishery

The Torres Strait Prawn Fishery (TSPF) is a separate and distinct fishery from both the Northern Prawn Fishery and the Queensland East Coast Trawl Fishery. Most vessels in the Torres Strait Prawn

fishery hold a Queensland east coast trawl endorsement, and some also hold a Northern Prawn Fishery endorsement. The Torres Strait Prawn Fishery is managed by input controls, including: limited entry, gear and vessel restrictions, individual effort limits, seasonal and area closures. The fleet is highly mobile and most vessels operate in this fishery on a part-time basis. Sixty-one vessels are licensed to fish the Torres Strait Prawn Fishery with up to 6,867 fishing days. In 2011, only ~1,300 days of fishing effort was recorded (Torres Strait Prawn Handbook 2012). Patterns of fishing effort in the Torres Strait Prawn Fishery have changed in recent years as a consequence of fuel costs and market prices for target species. Currently, brown tiger prawns dominate the catch (197 t), followed by blue endeavour prawns (72 t) and red-spot king prawns (4 t). This differs from average catches between 1991 and 2003, when endeavour prawns dominated the catch.

Relative abundance data since 1980 has been used in stock assessments, management strategy evaluations and fishery assessments for the Torres Strait Prawn Fishery (O'Neil and Turnbull 2006; Turnbull *et al.* 2009). As a result of the very low level of fishing effort in recent years, the annual catches of both tiger and endeavour prawns are now well below both the historic catch levels and the estimated Maximum Sustainable Yield (MSY) for tiger and endeavour prawns (Torres Strait Prawn Handbook 2012). The increase in tiger prawn catch per unit effort since 2000 is probably a consequence of the combined effect of fishers targeting tiger prawns in preference to endeavour prawns and a higher abundance of tiger prawns on the seabed. This is supported by the stock assessments conducted in 2004 and 2006, which are based on the monthly catch and standardised catch rates of tiger prawns since 1980. These assessments indicate that from 2002 to 2006 the tiger prawn biomass was increasing and higher than during the 1990's and the stock level required for maximum stock productivity (B_{msy}) (Torres Strait Prawn Handbook 2012).

Northern Prawn Fishery
The Northern Prawn Fishery extends from Cape Londonderry (Western Australia) eastwards to Cape York (Queensland). The Northern Prawn Fishery targets six different species of penaeid prawn, with brown and grooved tiger prawns a major sector of the fishery (Barwick 2010). The Northern Prawn Fishery is managed by the Australian Fisheries Management Authority through a combination of input controls including: limited entry (52 vessels), seasonal closures, permanent area closures, gear restrictions, and operational controls, which are all implemented under a Management Plan. In 2010, the banana prawn season commenced on the 31st March and concluded on the 10th June i.e., ten weeks, while the tiger prawn season commenced on the 1st August and concluded on the 29th November i.e., 17 weeks (Barwick 2010). In 2010, 52 vessels landed 1,628t of tiger prawns using 4,898 days of fishing effort. Annual monitoring surveys have occurred since 2002, in January and July, to provide independent data for recruitment and spawning indices.
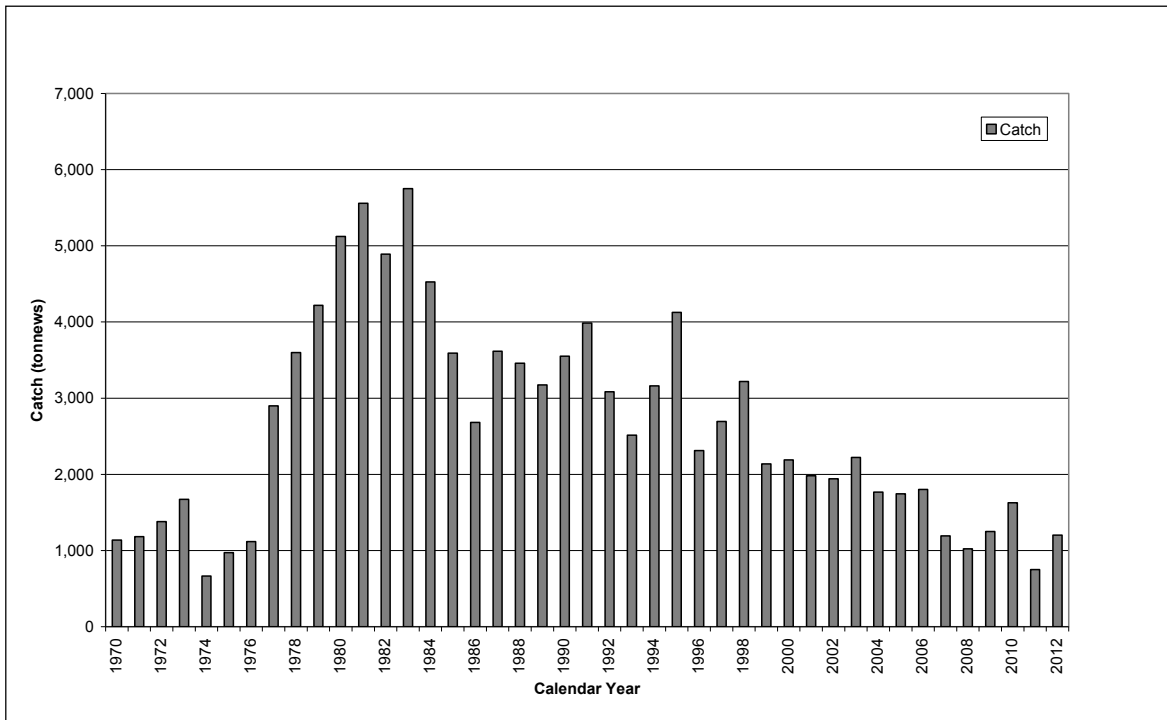
Figure 4.1. Tiger prawn commercial catch (tonnes) by year for the Northern Prawn Fishery (data from Barwick (2010) and AFMA (2013).
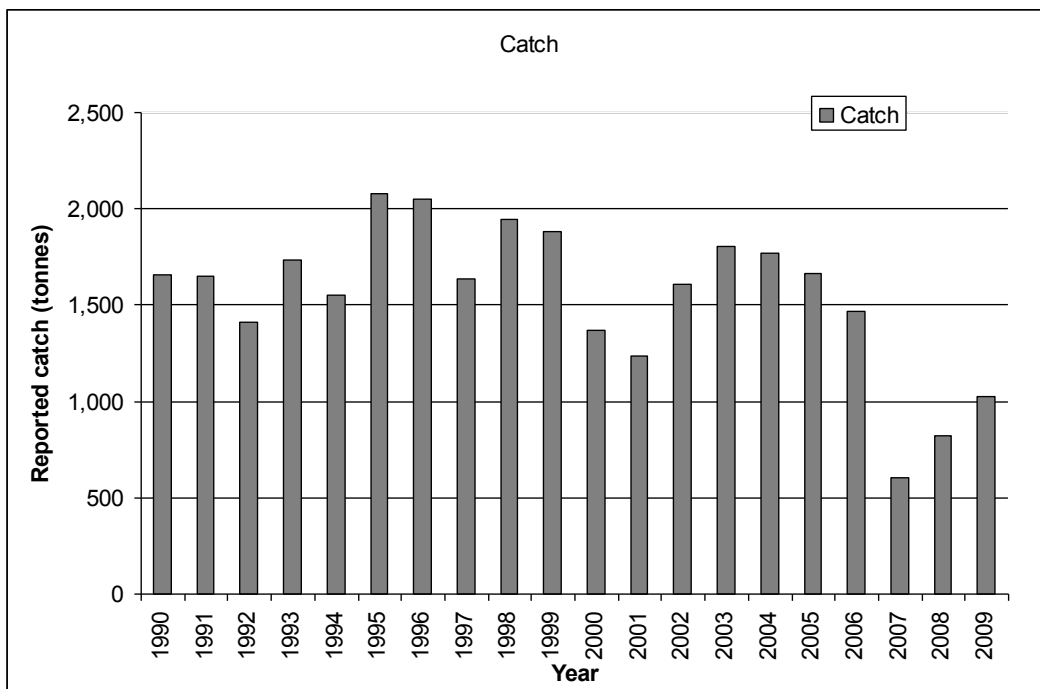
Figure 4.2. Tiger prawn commercial catch (tonnes) by year for the northern region of the Queensland East Coast Otter Trawl Fishery (data sourced from Fisheries Queensland).

Life history

Key points:
- Recruitment is probably affected by environmental factors but this has not been quantified.
- Juveniles are reliant on sea grass habitats as their primary habitat.

Life cycle, age and growth

Tiger prawns have a typical type-3 penaeid prawn life cycle (Dall et al. 1990), preferring relatively high salinity, sheltered inshore waters. Adults prefer habitats where sediments have a high (50-80%) mud content (Somers 1987a; Somers et al. 1987). Tiger prawns spawn in coastal waters up to 50m depth. The brown tiger prawn (*P. esculentus*) has variable spawning times and the spawning between August and October (i.e., Winter/Spring) provides the main source of recruits in the Torres Strait and Queensland east coast. However, in the Torres Strait, brown tiger prawns spawn throughout the year, with a second peak in spawning in summer that contributes a second (smaller) pulse of recruits that contribute to the Torres Strait Prawn Fishery. The grooved tiger prawn (*P. semisulcatus*) is thought to have two spawning periods, one in early summer and the other in autumn. Maturation and spawning of unablated female brown tiger prawns was favoured by conditions of warm temperature (26°C) and long days (14.5 h), whereas ovarian maturation did not occur at lower temperatures (20°C) and short days (12 h) (Crocos and Kerr 1983).

Pelagic larvae occur in high salinity water i.e., 30 to 35 ppt (Rothlisberg and Jackson 1987). Benthic post-larvae and juveniles have a strong association with seagrass habitats and algal beds (Young and Carpenter 1977; Young 1978; Coles and Lee Long 1985; Staples et al. 1985; Coles et al. 1987; Loneragan et al. 1994; O'Brien 1994a). In Torres Strait, juvenile tiger prawns are found on the tops of coral reef platforms, where they remain until they migrate into inter-reef areas as sub-adults. Tag-recapture studies in the Gulf of Carpentaria, Torres Strait, and Queensland east coast report movements by tiger prawns of generally less than 50 km (Kirkwood and Somers 1984; Somers and Kirkwood 1984); (Derbyshire et al. 1990; Watson and Turnbull 1993). This suggests that tiger prawns undertake limited migration between juvenile seagrass habitats and adjacent adult habitats.

Prawns are thought to have salinity and temperature optimums for biological processes such as growth and mortality e.g., (Saldanha and Achuthankutty 2000; Staples and Heales 1991), but these are likely to differ between species as a consequence of fundamental differences in their life history preferences (Dall et al. 1990). Laboratory studies on the brown tiger prawn (*P. esculentus*) found that after 50 days, juveniles could survive a wide range of temperature and salinity combinations (O'Brien 1994b). For example, survival of juvenile tiger prawns was >60% when water temperature was between 15 and 30°C and salinity was between 15 and 45 ppt. However, combinations of extreme temperature with extreme salinity were lethal. Fastest growth was estimated to occur at 30°C and a salinity of 30 ppt. O'Brien (1994b) concluded that juvenile brown tiger prawns were relatively euryhaline (i.e., able to tolerate a wide range of salinities) but were less tolerant of wide ranges in temperature, having impacts on growth rates, survival and distribution.

For grooved tiger prawns (*P. semisulcatus*), Xu *et al.* (1995) reported that salinity was negatively correlated with natural mortality estimates of a wild population in Kuwait waters i.e., the grooved tiger prawn had high mortality when salinity was low, due to flooding of the Shatt Al-Arab River.

The availability of **seagrass and algal bed habitats** preferred by post-larvae and juvenile tiger prawns may be affected by increases in: **sea level; tropical storm intensity & frequency; rain & flooding;** thereby impacting upon recruitment.

After ~3 weeks, **post larvae** settle in seagrass habitats and algal beds (Nov to Apr)

Currents can advect **larvae** to or away from suitable nursery habitats.

Juveniles spend ~1 to 2 months in seagrass or algal habitats

Eggs are benthic for 1 to 2 days, then hatch into planktonic **larvae** which prefer high salinity waters.

Sub-adults migrate to deeper waters adjacent to seagrass and algal habitats (Feb to Aug).

Adults migrate to **coastal** waters up to 50m deep. Spawning seasons varies: Spring & Summer for brown tigers and Spring & Autumn for grooved tigers. In southern Qld, only 1 spawning season in Spring. Not all spawning events provide recruits to the fisheries.

Temperature affects **growth rates** and **emergence behaviour**. Increases in temperature may enhance growth and subsequent survival as well as increase the catchability of tiger prawns.

Tiger prawns mature into **adults** at ~6 to 12 months of age.

Changes in pH may alter carbonate saturation and therefore the **moult** and **growth** of all stages of the lifecycle. Impacts unknown.

Figure 4.3. Generalised life cycle of tiger prawns. (Images sourced from QDAFF).

Distribution, habitat and environmental preferences

The brown tiger prawn (*P. esculentus*) is endemic to coastal waters of tropical and sub-tropical Australia, and can be found in waters up to 50 m deep (Kirkegaard and Walker 1969; Racek and Dall 1965). It is likely that there are separate stocks of brown tiger prawns on the east and west coast of Australia (Courtney 1997).

The grooved tiger prawn (*P. semisulcatus*) is a tropical species and is more widespread in its distribution than the brown tiger prawn, occurring in coastal waters of the Indian and western Pacific oceans, where it is trawled in waters up 130 m deep (Grey et al. 1983; Kailola et al. 1993). The benthic post-larvae and juveniles of both species of tiger prawn prefer seagrass and algal bed habitats. Adults of the brown tiger prefer habitats with coarse, sandy sediments, while adults of the grooved tiger prawn prefer habitats with a high (50-80%) mud content (Somers 1987b; Somers et al. 1987).

In a recent detailed study of the influence of environmental parameters on standardised catch rates of brown tiger prawns in Moreton Bay, Courtney *et al.* (2011) found that flow of the Brisbane River had a significant (but small) negative impact, as would be expected from life history information and

previous studies. This is in contrast to that reported from a broadscale analysis of relationships between fisheries catch and climate parameters, where tiger prawn catches along the east coast of Cape York were positively related to rainfall and the Southern Oscillation Index (Meynecke and Lee 2011). A positive correlation between catch (adjusted for effort) and SOI suggests that increased catches occur when the SOI is positive (indicative of La Nina events and increased rainfall).

Cool water is thought to be the major factor that restricts the distribution of *Penaeus esculentus* in Australia (O'Brien 1994b).
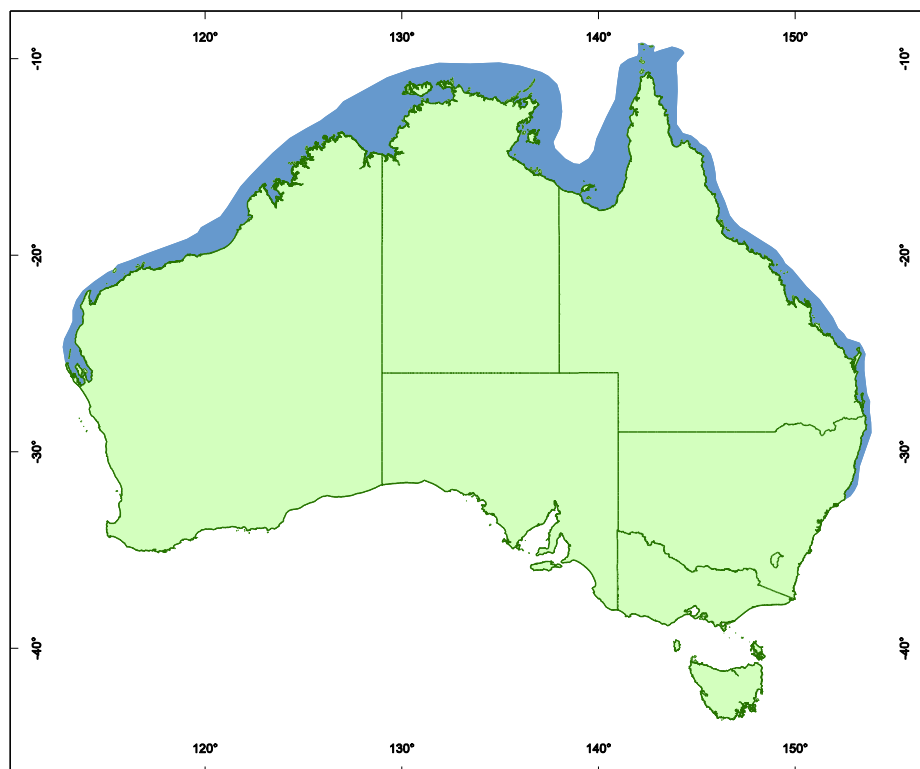
Figure 4.4. Australian distribution of tiger prawn.

Predators and prey
Tiger prawns eat a variety of organisms including bivalves, gastropods, ophiuroids, crustaceans and polychaetes (Wassenberg and Hill 1987). Tiger prawns are eaten by a variety of predatory fish.

Recruitment
Flood *et al.* (2010) stated that variable recruitment of brown tiger prawns in Torres Strait was influenced by environmental factors. Fishers operating in the Torres Strait Prawn fishery anecdotally report that a dry preceding year favours tiger prawn recruitment while a wet preceding year favours endeavour prawn recruitment (C. Turnbull pers. comm.). Recruitment was impacted by cyclones in Western Australia, with the effect depending on timing (Penn and Caputi 1986a). Cyclones and associated rainfall early in the wet season reduced salinity of near shore juvenile habitats and reduced recruitment, whilst cyclones later in the wet season increased the turbidity of water and probably reduced predation of sub-adult and adult tiger prawns that had moved away from shallow nursery habitats.

Current impacts of climate change

Key points:
- There are known links between climate factors and tiger prawn population dynamics.
- Effects are likely to vary regionally – temperature should be an important factor towards the southern limit of the distribution of brown tiger prawns.

There are no documented impacts of climate change on tiger prawns. Tiger prawns utilise inshore areas during their life cycle. These species are particularly dependent on seagrass and algal beds as nursery habitats. Primary and secondary climatic factors that impact (positively or negatively) upon the abundance, distribution and quality of these habitats will impact tiger prawns. Impacts include elevated sea surface temperatures (postulated positive effects on prawn growth and survival, emergence duration and exposure to capture and productivity of seagrass habitats); sea level rise; tropical storm activity. Tiger prawns are also exposed to changes in ocean pH, and like other species it is uncertain whether prawns in general and tiger prawns as individual species are sensitive to the anticipated change in ocean pH and its effect on calcium carbonate formation and moulting.

Meynecke and Lee (2011) report significant positive correlations between tiger prawn catch adjusted for effort in select regions of Queensland with rainfall and SOI. Relationships between tiger prawn catch (adjusted for effort) and wet season rainfall were not significant ($r<0.30$, $p>0.05$) in the five northern regional areas where the majority of tiger prawns are harvested. However, catch adjusted for effort was significantly correlated to sea surface temperature in two of the five northern regional areas that account for most of the tiger prawn catch (see Fig. 3 of Meynecke and Lee 2011).

Sensitivity to change

Key points:
- Tiger prawns are sensitive to changes in the seagrass and algal beds as these are the primary habitat of juveniles.
- Tiger prawns appear to have a wide tolerance for temperature and salinity.

Tiger prawns are dependent on seagrass and algal beds as juveniles and would be sensitive to any changes in the abundance, quality or distribution of this habitat. For example, tropical storm activity (i.e., severe cyclones) negatively impacted juvenile tiger prawn habitats in Western Australia and tiger prawn recruitment was reduced in the two years subsequent to the cyclone (Penn and Caputi 1986b).

Resilience to change

Key points:
- Tiger prawns are possibly less resilient to change than other prawn species because of their dependence on specific habitats as juveniles.

Brown *et al.* (2010) simulated the effects climate change may have on primary production in Australian marine ecosystems using the food web model Ecosim. They predicted that in the Gulf of Carpentaria, tiger prawn abundance would decline in response to high predation rates or strong competition with other functional groups.

Morison and Pears (2012) completed an expert based vulnerability assessment of the Queensland East Coast Otter Trawl Fishery and concluded that tiger prawns in Queensland had a medium level of ecological vulnerability to ocean acidification and its consequences for moult success and therefore recruitment. They also found a medium level of ecological vulnerability to higher sea surface temperature, sea level rise, changed rainfall patterns and increased tropical storm activity.

Water temperature is thought to be a major factor restricting the distribution of the brown tiger prawn (*P. esculentus*) in Australia (O'Brien 1994b). As water temperatures increase, brown tiger prawns might increase their distribution southerly (i.e., into New South Wales) and become more abundant in areas where seagrass occurs, but temperature limits the survival of brown tiger prawns.

Other
Ecosystem level interactions
Tiger prawns are a key food source for many estuarine-dependent and inshore coastal species. Changes in the distribution and abundance of tiger prawns may potentially affect the distribution and abundance of their predators. Tiger prawns are also dependent on specific habitats (sea-grass and algal beds) as juveniles. Changes in the distribution and abundance of these habitats may also affect tiger prawn populations.

Additional (multiple) stressors
Tiger prawns are exploited by trawl fishers, although the main trawl fisheries are managed such that tiger prawn stocks in northern Australia are not thought to be over-exploited (Punt et al. 2010).

Critical data gaps and level of uncertainty
Knowledge of the impact of many physical factors on the population dynamics of each species of tiger prawn is limited. Therefore, the impacts of climate change on these species are mostly speculative and uncertain.

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5. Mud crab, giant (*Scylla serrata*) and orange (*S. olivacea)*

Authors: Emily Lawson, Thor Saunders, and Julie Robins

The fishery

> **Key points:**
> - Mud crabs are harvested at substantial levels by all sectors across the majority of northern Australia.
> - There is a male only harvest fishery in Queensland waters; elsewhere males and females may be harvested.
> - In northern regions catch is influenced by SOI and rainfall/riverflow, while temperature is likely to play a more significant role in southern regions.

Western Australia

There is only a small developmental fishery for the orange mud crab, *Scylla olivacea*, in Western Australia. Currently annual catches are less than 5 t (Department of Fisheries 2011).

Northern Territory

The mud crab fishery is one of the key Northern Territory (NT) managed wild harvest fisheries. Approximately 400 tonnes of mud crabs were caught in the 2010 commercial wild harvest sector, down from over 1000 tonnes in 2001 (Figure 5.1). Two species of mud crabs are found in NT waters; the giant mud crab (*S. serrata*) accounts for 99% of the catch from all sectors, while *S. olivacea* constitutes the remainder (Northern Territory Government 2011). Crabbing operations are confined to coastal and estuarine areas, predominantly on mud flats, with most activity concentrated in the Gulf of Carpentaria. The estimated gross value of the catch was $8 million in 2010 (Northern Territory Government 2011).

Parallel surveys in 2000-01 highlighted the importance of the mud crab resource to recreational (including Fishing Tour Operators) and Indigenous fishers who were estimated to harvest 82,000 and 86,500 crabs, respectively, with a combined weight of about 135 tonnes, during a 12-month period (Henry and Lyle 2003).

Both male and female mud crabs can be retained in the Northern Territory. Rules and regulations apply to each fishing sector, such as minimum legal size, possession limits, gear restrictions and no harvest of berried females (i.e., with eggs attached) or newly moulted 'soft' crabs.
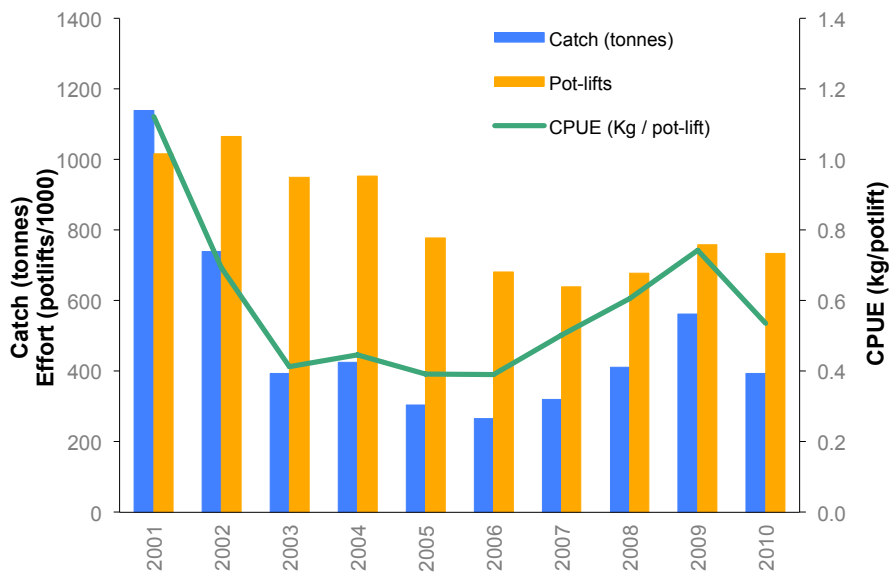
Figure 5.1. Catch per unit effort (CPUE) for the Northern Territory commercial Mud Crab Fishery from 2001 to 2010.

Queensland

The Queensland mud crab fishery primarily targets *S. serrata*, although *S. olivacea* occurs in Queensland waters and comprises a small component of the catch (Jebreen *et al*. 2002). The commercial sector accounts for ~59% of the total mud crab harvest, while the recreational sector accounts for about 40%, and the Indigenous sector <1% (Fisheries Queensland 2010). Pots and hoop dillies are the primary means of capture, with 'hooking' prohibited since 1995. Crabbing can be carried out in association with other forms of fishing, such that pots are set and left while undertaking other activities like netting (commercial) or line fishing (recreational or commercial). Mud crabs are also caught as bycatch in the Queensland set gill net fishery.

The Queensland commercial mud crab fishery is managed by input controls including: restricted commercial entry, limits on the number and types of pots (maximum 50), minimum legal size (150 mm carapace width), a male only fishery (females protected), spatial closures, and a possession limit of 10 for the recreational fishery.

In 2010, the reported commercial catch was 1,192 tonnes (Figure 5.2), with 1,015 tonnes taken from the Queensland east coast and the remainder from the Gulf of Carpentaria (Fisheries Queensland 2011). The gross value of production for the commercial fishery was in the order of $19 million, with 375 out of a possible 437 licences reporting landings of mud crab (Fisheries Queensland 2011).

Fisheries for mud crabs are associated mostly with estuaries. However, in Queensland, some of the major commercial mud crab areas are not rivers, but large sheltered areas behind islands e.g., southern Moreton Bay, Great Sandy Straits, the Narrows near Gladstone, Broadsound north of Rockhampton, and Hinchinbrook Channel north of Townsville (Fisheries Queensland 2011). The duration of the main peak fishing season increases with latitude, being eight months in north Queensland and ten months in Moreton Bay (Hill 1982). Main landings occur between December

and June and are related to water temperature, as activity and feeding are reduced at temperatures below 20°C (Hill 1980).
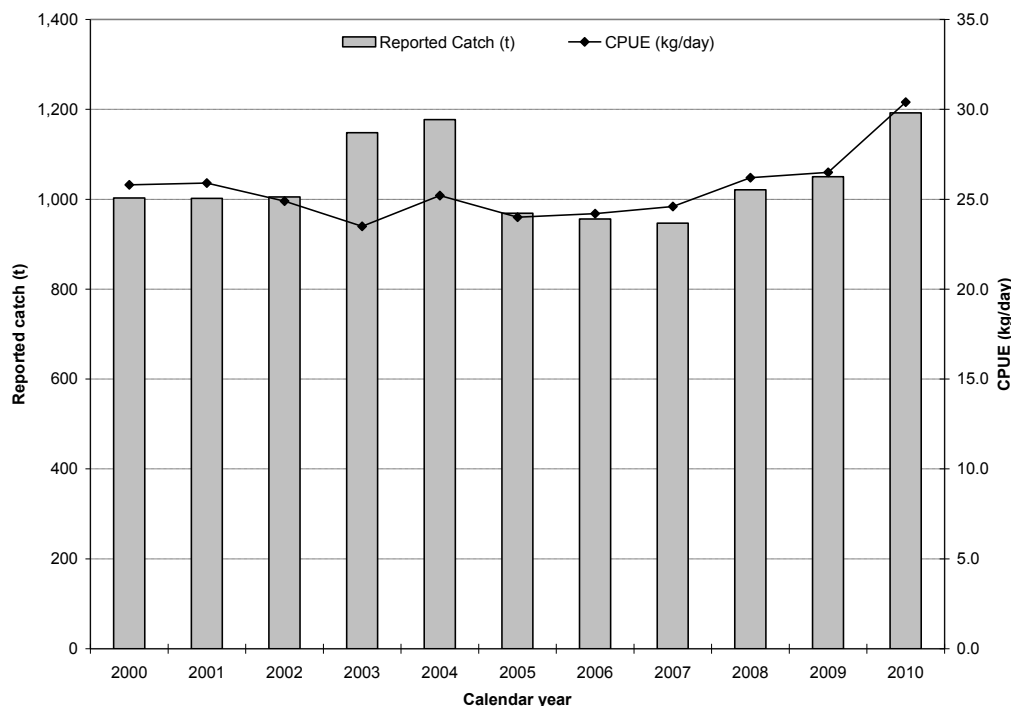
Figure 5.2. Mud crab commercial catch (tonnes) and catch per unit effort (CPUE) by year for Queensland, east coast and Gulf of Carpentaria pooled (Source: Fisheries Queensland Annual Status Report 2011 Mud Crab Fishery).

Relationships between catch and environmental variables

There is variable evidence as to the importance of different environmental factors on the catch rate of mud crabs (Table 5.1). Williams and Hill (1982) conducted fishery independent sampling and found that catches were not correlated with salinities (range of 24 to 35 ppt; $r = 0.09$, n=44), but were significantly correlated with daily water temperature ($r = 0.56$, n=44). Williams and Hill (1982) found that the catchability of mud crabs was (negatively) influenced by low water temperatures in winter as well as the moulting cycle. Moulting mostly occurs in October and to a lesser extent in December and January. Catchability also varied with the size and sex of mud crabs.

Helmke *et al.* (1998) reported that Queensland commercial crabbers generally believed that environmental factors were responsible for declining catches in the Gulf of Carpentaria in 1998. The crabbers believed that a drop in the number of crabs caught was a consequence of: (i) 'the long period of rain early in the year and an extended period of freshwater runoff', when adult crabs may have been flushed out into the Gulf of Carpentaria and then tried to return to successive estuaries as they moved down the coast; and (ii) recruitment failure caused by high rainfall two years' previous. Helmke *et al.* (1998) also reported a low percentage of tag-recaptures in the Weipa region during an

intensive tag-recapture study and suggested that this was indicative of a high migration rate possibly linked to the long period of freshwater runoff 'that local crabbers believe cause crabs to move'.

Loneragan and Bunn (1999) reported significant positive correlations between (fishery-dependent) mud crab catch and summer freshwater flow in the Logan River estuary (a sub-tropical estuary), while Robins *et al.* (2005) and Halliday and Robins (2007) found significant correlations between fishery-dependent mud crab catch and flow and rainfall for the Fitzroy and Port Curtis regions (near the Tropic of Capricorn). Robins *et al*. (2005) reported a positive correlation between the catch of mud crabs and summer freshwater flow. Loneragan and Bunn (1999) suggested that the downstream migration of adults as a consequence of freshwater flow may enhance the catchability of adults by moving them to fishable areas as well as enhance the survival of juveniles (i.e., a recruitment effect) by reducing competition for burrows and any cannibalism, potentially increasing the overall abundance of the species.

More recently, Meynecke *et al.* (2012) used fishery dependent catch data and reported significant regional correlations between monthly mud crab catch per unit effort, mean seasonal flow and mean summer sea surface temperature.

Life history

- Mud crabs spawn in marine waters during Spring and Summer and are highly fecund.
- Mud crabs grow quickly and attain maturity after ~18 months in the tropics but can take up to 36 months in the sub-tropics (eg. Moreton Bay, SE Queensland).
- Growth rates decrease with increasing latitude, while size-at-maturity increases with increasing latitude. These effects are likely to be temperature related.
- Currents, salinity, temperature, food supply and predation are more likely to influence recruitment than the abundance of spawning females.

Life cycle

The life cycle of the mud crab involves several stages that utilise both marine and estuarine environments (Arriola 1940). The mating process begins when a mature, hard shelled male finds a female that is ready to moult (presumably through the release of pheromones from the female). The male then carries the female with his first pair of walking legs for a period of three to four days before she moults and is subsequently inseminated (Ong 1966). The crabs remain "doubled" until such time as the females' shell has hardened, typically about five days (Perrine 1978). The female stores the spermatophores for between two and seven months (Ong 1966), during which time up to three batches of eggs can be fertilised (Heasman *et al*. 1983). The eggs (which may number from two to 11 million; Davis, 2004) are later extruded onto the ventral surface of the females' abdomen where they remain until hatching.

Once mated, the females can migrate up to 50 km offshore into waters 20 to 40 m deep (Hill, 1994). The peak of the spawning event (comprising both migration and hatching) in Australia, occurs from September to November in the tropics, and from October to December in the subtropics (Heasman *et al*. 1985; Knuckey 1999). The timing of the migration (i.e., before the monsoon season), suggests that migration is not triggered by low salinities in estuaries (Hill 1994), although both Quinn and

Kojis (1987) and Heasman *et al.* (1985) suggested that peak spawning in the tropics does indeed coincide with periods of high nutrient input associated with monsoonal rainfall.

The rate of egg and larval development in *Scylla* species is inversely proportional to temperature, with an optimal temperature of 29°C (Hamasaki 2003). Salinity also appears to be critically important with salinities below 20 ppt resulting in high mortality rates (Hill 1974; Quinn and Kojis 1987; Nurdiani and Zeng 2007), however, experimental aquaculture work indicates that juvenile mud crabs can be reared at salinities within 10 to 25 ppt, provided temperature is greater than 25°C (Ruscoe *et al.* 2004). Hatching occurs about 12 days post extrusion, after which the planktonic zoea pass through five discrete stages over the next 12 to 15 days (Brown 1993).

The megalopae are semi-pelagic bottom-dwelling (Fielder and Heasman 1978) and after five to 12 days metamorphose into juvenile crabs (Williams 2002). Webley and Connolly (2007) proposed that megalopae settle on the coastal shelf, possibly near river mouths and then move along the substrate as they migrate upstream towards mangrove and seagrass habitats (Webley *et al.* 2009). Stage one crabs are only about 4 mm wide and have rarely been seen in the wild, but frequent moults mean that they grow very quickly.

Age and growth

Mud crabs grow through a series of moults and (unlike fishes) do not retain hard parts suitable for ageing. Hence, estimates of mud crab age are crude and rely on cohort analysis, which infer a maximum life span of three to four years (Heasman 1980).

Mud crabs grow quickly, reaching 80 to 100 mm carapace width (CW) in their first year; 130 to 160 mm CW in their second year, and around 200 mm CW (under ideal conditions) in their third year (Heasman 1980). Both growth rates and the size at first maturity vary with latitude (Fielder and Heasman 1978). Mud crabs in Australia reach maturity in 18 months in the tropics, but can take up to 36 months to reach maturity in the sub-tropics such as Moreton Bay. The minimum size at maturity is larger in sub-tropical areas than tropical areas (Brown 1993).

Table 5.1. Summary of studies correlating mud crab catch with environmental variables

Reference and data source	Analysis	Result	Discussion points
Williams & Hill, 1982 • Fishery independent	Pearson correlation for catch vs salinity/daily temperature	No correlation with salinity ($r = 0.09$) but moderate positive correlation with daily water temperature ($r = 0.56$)	Catch rates related to water temperature but not salinity
Loneragan & Bunn, 1999 • Fishery dependent	Pearson correlation for Log catch vs Log seasonal river flow (Logan River)	Catch strongly correlated ($r = 0.9$) with summer river flow only	Catch rates related to summer river flow
Manson et al., 2005 • Fishery dependent	Multiple regression analysis of catch rates vs mangrove area and perimeter.	Moderate and significant positive relationship between mangrove perimeter and catch rates ($r^2 = 0.53$)	Catch rates related to mangrove perimeter
Robins et al., 2005 • Fishery dependent	Pearson correlation and all subsets regression for concurrent and lagged (i.e. 1 or 2 years) Log catch vs Log seasonal river flow/ rainfall (Fitzroy region). Two data sets used: 1960-1980 from QLD Fish Board (QFB) and 1988-2002 from Fisheries Queensland commercial logbook catch data.	Ambiguous results from analyses of QFB data – some significant positive and negative relationships. Significant positive relationship ($r^2 = 0.7$) in Fisheries Queensland commercial logbook catch data between Autumn catch lagged by two years and river flow. All subset regression identified several alternate models that explained between 70 and 97% of the variation in mud crab catch for the QFB and CFISH data respectively.	The QFB data was not adjusted for effort and so inferences from subsequent analyses are limited. The Fisheries Queensland commercial logbook catch was corrected for effort, with model outputs similar to those given in Loneragan & Bunn (1999).
Halliday & Robins, 2007 • Fishery dependent	Pearson correlation and all subsets regression for concurrent and lagged (i.e. 1 or 2 years) Log catch vs Log seasonal river flow/ rainfall (Port Curtis region). Two data sets	Significant correlations between QFB catch data and summer and autumn flows. Flows lagged by 1 and 2 years were also significantly (negatively) correlated with catch. Four alternate GLMs identified explaining between	Evidence of freshwater flows in autumn affecting the recruitment strength of mud crabs, concurring with the suggestion of

Meynecke and Lee 2011 • Fishery dependent	used: 1960-1980 from QLD Fish Board (QFB) and 1988-2002 from Fisheries Queensland commercial logbook catch data. Linear regression of wet season and monthly catch adjusted for effort (CAE) against Sea Surface Temperature and Rainfall for Queensland divided into 8 regional areas.	60% and 66% of variability (i.e., adj. R^2) of the QFB catch. Two significant alternate models (adj. R^2 71% and 63%) for the commercial logbook catch, both containing spring flow or rain lagged by two years. Significant positive correlation between wet season CAE and rainfall for 2 of 8 regions. Significant GLMs explaining 43% to 68% of variation in annual CAE for 3 regions; of which 2 regions had SOI (May to Oct) and annual rainfall. Significant GLM's explaining 22% to 71% of variation in CAE lagged by two years. Temperature important in 1 region, wet season rainfall important in the other 2 regions.	recruitment effects by Loneragan & Bunn (1999). Suggested significant relationships reflected reduction in the numbers of sub adult and adult crabs in river systems from increased runoff, thus enhancing survival of juveniles. Relationships variable between regions.
Meynecke *et al*. 2012 • Fishery dependent	Monthly catch and effort data (CPUE) analysed by forward linear regression of 29 catchments with 52 river systems across the Northern Territory, Queensland and New South Wales. Used multi-dimensional scaling to describe similarities in annual CPUE, rainfall, temperature and flow.	Significant relationships between seasonal rainfall lagged by one season and annual maximum SOI with NT CPUE for 4 of 7 river systems, with Qld CPUE for 6 of 14 river systems. For NSW CPUE, temperature was significantly related to the most influential climate parameter. MDS showed regional groupings in all states.	Identified regional patterns in mud crab catches. SOI was a good predictor of CPUE in Qld, NT and some parts of NSW. CPUE significantly related to rainfall and or flow in many rivers examined. Monthly SST and CPUE related in some areas.

Distribution, habitat and environmental preferences

Mud crabs of the genus *Scylla* commonly occur throughout tropical to warm temperate areas of the west Pacific and Indian Oceans (Keenan 1999). Their distribution encompasses the Asian sub-continent and Japan, northern and eastern Australia and from the east coast of Africa across to Tahiti (Ryan 2003). In Australia, they inhabit regions extending from Exmouth Gulf on the coast of Western Australia, through to the Northern Territory and Queensland to the southern coast of New South Wales (Figure 5.3; Knuckey 1999).

Mud crabs usually inhabit estuarine channels, sheltered coastal habitats and shallow tidal flats associated with mangrove communities (Hill *et al*. 1982). Juveniles usually remain in the intertidal zone (amongst mangroves), whereas adults tend to be more abundant in the sub-tidal zone (Hill *et al*. 1982).

Predators and prey

Mud crabs form an important part of mud flat and mangrove food webs as they consume a variety of organisms while they are prey for large teleost fishes, rays, sharks, turtles and crocodiles (Poovachiranon 1992). Isotope studies of the stomach contents of mud crabs suggest that their diet changes with size. Small mud crabs (i.e., 60 to 99 mm CW) are omnivorous, feeding on small crabs and plant material, while medium (100 to 139 mm CW) and large mud crabs (140 to 179 mm CW) are predominately carnivorous, feeding on slow moving invertebrates such as grapsid crabs, prawns, molluscs, worms and some fish (Thimdee *et al.* 2001). Mud crabs are generally nocturnal feeders, emerging from their intertidal burrows at dusk, moving slowly over the substrate to capture prey and to scavenge before returning to a burrow by dawn (NSW DPI 2008). They have a home range of approximately 500 m, and use their larger claw for crushing while the smaller claw is used for biting, cutting and manipulating the prey (Ryan 2003). Their feeding activity depends on environmental factors such as temperature and physiological factors such as moult condition.

Recruitment

Mud crabs are highly fecund and it is likely that factors such as currents, salinity, temperature, food supply, suitable habitat and predation are more important in determining recruitment levels than the number of spawning females (Ian Brown pers. comm.). Halliday and Robins (2007) reported significant correlations between mud crab catch (adjusted for effort) and autumn or spring flows two years prior to the catch in the Fitzroy and Port Curtis regions of central Queensland. These results suggest an effect of freshwater flows on successful recruitment to the estuary and concur with the hypothesised recruitment effects by Loneragan and Bunn (1999). It is unlikely that it is the flow per se that increases recruitment, as mud crab early life stages are not tolerant of low salinity water. More likely is the effect freshwater flows have on nutrient inputs and productivity of the estuary, leading to enhanced food opportunities and faster growth rates, which may lead to greater survival of young crabs.
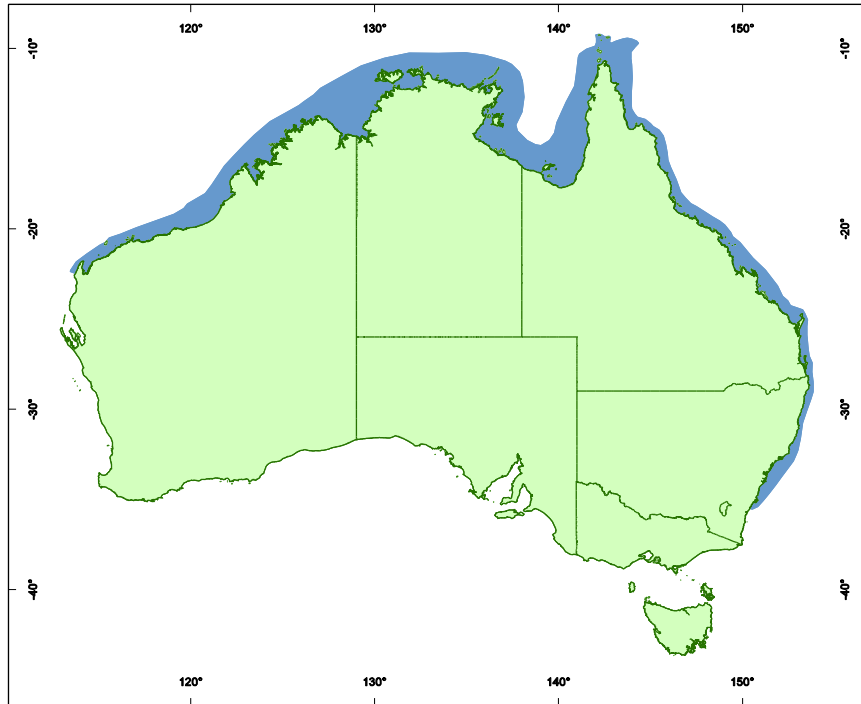
Figure 5.3. Australian distribution of mud crab
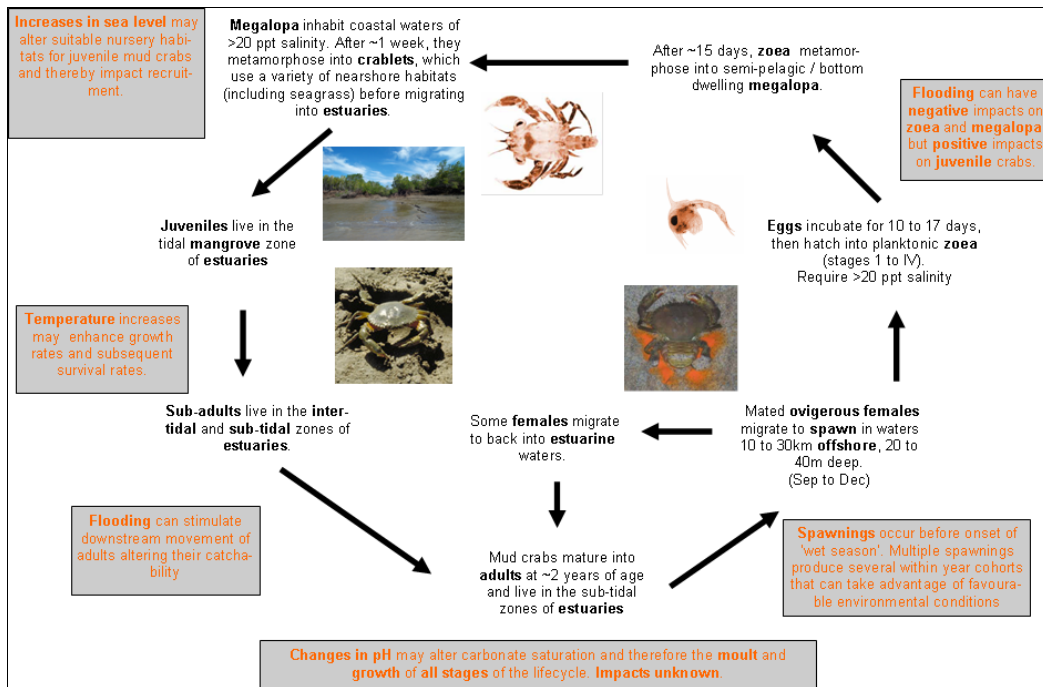
Figure 5.4. Generalised life cycle of mud crab (Images courtesy of DAFF, NTDoR and GBRMPA)

Current impacts of climate change

- There are no known current impacts on mud crabs attributable to climate change.

Rainfall and temperature are suggested to be important potential environmental drivers affecting mud crab catchability, growth rates and recruitment success. Notwithstanding this there are no current impacts on mud crabs that can be attributed to climate change.

Sensitivity to change

- Rainfall/riverflow and temperature appear to have a significant influence on mud crab catchability (and possibly juvenile survival) and growth.
- Mud crabs are likely to be sensitive to changes in the distribution of estuaries and associated wetland habitats, including mangroves and saltmarshes, with sea level rise.

Mud crabs are widely distributed throughout the tropics and sub-tropics of the west Pacific and Indian Oceans. Several aspects of the life cycle of mud crabs are sensitive to temperature, including survival and development of early life stages (Hamasaki 2003; Ruscoe *et al.* 2004), growth rates (Fielder and Heasman 1978; Brown 1993) and feeding activity (Williams and Hill 1982). To date, studies have focused on the lower temperature tolerances and no studies have examined the sensitivity of mud crabs to temperatures above 35°C. Mud crabs are also sensitivity to salinity (Ruscoe *et al.* 2004), with early life history stages requiring >20 ppt. Mud crabs utilitise estuaries and associated wetland habitats, including mangroves and saltmarshes, and will be sensitive to changes in the distribution of these habitats that may occur if sea level rises significantly. In addition, mud crab abundance will be sensitive to any changes in the Southern Oscillation Index (SOI) as between 30 and 40% of the catch variability is explained by La Niña phases which are associated with increased rainfall and higher temperatures in northern Australia (Meynecke *et al*. 2010).

Other physical drivers that influence the abundance of mud crabs arecurrents, tides and wind that control dispersal and settlement during the larval stage the lunar cycle that affects the timing of moulting and migration. Additionally the mangroves and mudflats inhabited by mud crabs are often rich in organic material and microorganisms, thereby having a high biochemical oxygen demand. The shallow water of these areas with ebbing or flooding tides is likely to contain low levels of dissolved oxygen. Therefore, mud crabs may be subject to hypoxic stress (Davenport and Wong 1987), especially during high temperatures.

Resilience to change

- Mud crabs appear to be resilient to a wide range of environmental conditions throughout their life history.

The resilience of mud crabs to long-term changes in the climate regime is unknown. However, there are many aspects of the life history of mud crabs that suggest they may be resilient to change, at least in the short term (i.e., <50 years). Mud crabs have a wide geographic distribution (including a large latitudinal range), are an omnivorous detritivore capable of using a wide variety of prey items,

inhabit a wide range of estuarine habitats, have extended spawning seasons, and high fecundity, with short time to maturity. Further research is needed to determine if the early life history stages of mud crabs may be critically limited by altered water pH and what is their upper temperature tolerance.

Other

- Mud crabs provide a source of food to higher-level predators.
- The effect of specific environmental factors, including water extraction, on each stage of the mud crab life cycle is poorly understood.

Ecosystem level interactions

Biological drivers can include predation by other crabs (i.e., cannibalism) or through predators like crocodiles, sharks, rays, fish, dingoes and humans. In addition, the distribution, abundance and diversity of estuarine habitats such as mangroves, mudflats and seagrasses are likely to be related to the abundance of mud crabs (Meynecke *et al.* 2007; Webley *et al*. 2009).

Additional (multiple) stressors

Both the market demand and catch rate of mud crabs have increased substantially over the past decade but there has also been large variations in catch, particularly in the Gulf of Carpentaria and the Northern Territory. Fluctuations in catch rates greater than a factor of eight were thought to be driven by climate parameters and are likely to increase further with climate change. Such variations may pose a challenge to the viability of the commercial fishery.

The years 2000 and 2001 saw record mud crab (*Scylla serrata*) catches throughout its range in Australia, presumably due to a combination of high fishing effort and favourable recruitment in the preceding years. This peak was followed by a significant decrease in catch in all relevant jurisdictions, with the magnitude of the decline greatest in the Northern Territory. These large commercial catches are occurring again in 2011 (Fisheries Queensland commercial logbook data).

Heavy rainfall events (causing major flooding) can have an associated dieback of seagrass beds. Seagrass has been proposed (but not validated) as a preferred habitat for crablet colonisation, prior to their movement into estuaries (Webley *et al*. 2009). Therefore, negative impacts of climate change on seagrass beds may subsequently impact on the recruitment success of mud crabs at their crablet stage.

Critical data gaps and level of uncertainty

More field studies to examine the linkages between specific environmental drivers, habitat mosaics and their influence on the mud crab life cycle are needed. There is little evidence of cause and effect which is an issue with most ecological/climate modelling. This includes the definition of activity and spawning trigger values, such as tide, temperature and salinity, for a number of Australian mud crab populations from the various biogeographic regions.

Despite the continued use of freshwater resources and subsequent alteration of the quantity and quality of freshwater flowing down rivers to estuaries, there is limited understanding of the impacts of such changes on estuarine flora and fauna. To understand the impact of changed flow, we need

first to understand and quantify the role of freshwater on populations of estuarine-dependent species.

Acknowledgements

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6. Saucer scallop, *Amusium japonicum balloti*, and mud scallop, *A. pleuronectes*

Author: Julie Robins

Queensland saucer scallop, *Amusium japonicum balloti*. (Image sourced from QDAFF).

The fishery

> **Key points:**
> * Harvest is variable and probably linked to environmental drivers.
> * The fishery is dependent on scallops that are one to two years old.

Queensland east coast

Two species of scallop are harvested along the Queensland east coast and, unlike other scallop species in southern Australia, are harvested by trawl methods rather than dredge. Scallops are harvested by trawlers in the East Coast Otter Trawl Fishery, which operates from the border between Queensland and New South Wales northwards to the Torres Strait. The saucer scallop, *Amusium japonicum balloti,* is the main species harvested in Queensland waters south of 20°S while the mud scallop, *A. pleuronectes*, is the main scallop species harvested in waters north of 20°S and is generally regarded as a by-product species (Williams 2002).

Although scallops form part of the multi-species East Coast Otter Trawl Fishery, there are specific management restrictions and gear differences that separate the scallop sector from the prawn sectors. The Queensland East Coast Otter Trawl Fishery is managed by input controls including: limited entry, individual effort limits, vessel restrictions, and spatial and temporal closures. In

addition, for saucer scallops, there are minimum legal shell size limits, as well as rotational spatial closures and temporal closures that are designed to maintain broodstock levels and maximise harvest rates.

The annual harvest of Queensland saucer scallops is highly variable (Figure 6.1). Monthly catch rates of saucer scallops are also highly variable, peaking in January and November when spatial and temporal closes (respectively) are opened to fishing (Campbell *et al*. 2010).
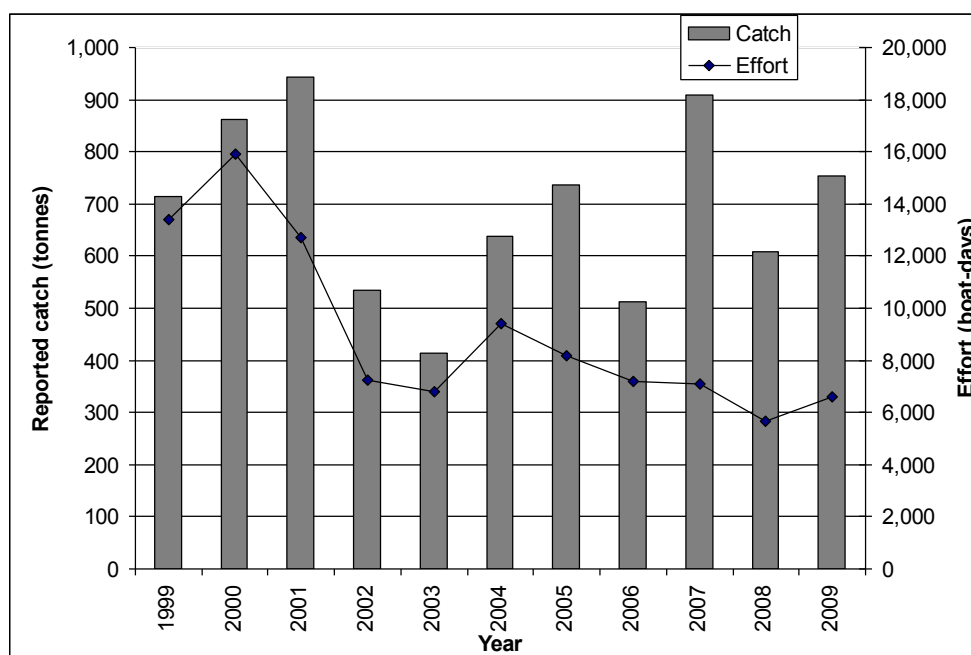
Figure 6.1. Scallop commercial catch (tonnes) and effort (boat days) by year for the Queensland East Coast otter Trawl Fishery (data source Fisheries Queensland)

Life history

> **Key points:**
> * Recruitment is highly variable in Queensland and Western Australian stocks.
> * Recruitment variability in WA is linked to a weak Leeuwin Current that occurs in El Nino years.
> * Recruitment in Queensland is speculated to be linked to the East Australian Current.

Life cycle, age and growth

The saucer scallop has a life history that is typical of Pectinid bivalves. Adults spawn in a single spawning season from winter to spring. Saucer scallops spawn eggs and sperm that are fertilized in the water column. After a short incubation period (<2 days), eggs hatch into a two-stage larval phase first becoming a trochophore (~28 h after fertilization), then a veliger (2 days after fertilization), both of which are pelagic (Rose et al. 1988). The pelagic larval phases last up to about three weeks (Rose et al. 1988) and during this time larvae can freely swim. Laboratory experiments have shown that

saucer scallop larvae require water temperatures <25°C (Cropp 1993). The distribution of pelagic larval scallops and the subsequent recruitment of scallop spat to productive seabed areas in speculated to the influenced by water currents and other hydrological features such as cyclonic eddies.

The pelagic veliger stage metamorphoses into a pediveliger stage at around 20 days, developing a ciliated foot and a distinct byssal gland that in other scallop species produces byssus threads for attaching the spat to hard surfaces. Saucer scallops have a limited byssal stage (Rose et al. 1988). . Spatfall (the settling of larval scallops to the sea floor) and survival of spat is probably important in determining the productivity of sea bed areas for adult scallops. Successful spatfall of other species of scallop requires suitable substrates for settlement and is negatively affected by shifting sands on the seafloor. Whether this is the case for saucer scallops is unknown, but their short byssal phase may make them more tolerant of shifting sands. Once settled on the seafloor, saucer scallops are thought to be relatively sedentary i.e., they do not undertake migration.

Saucer scallops have the best swimming ability of pectinid bivalves and can swim up to 20 m in one burst. Juvenile saucer scallops grow quickly, up to 2.2 mm per week (Joll 1988), reaching 90mm shell height in 33 to 42 weeks (Williams and Dredge 1981), and participate in their first spawning at between nine and 12 months of age (Dredge 1981). Saucer scallops are dioecious i.e., separate sexes. They live to about three years of age, although most do not survive to this age because of high natural mortality rates (Dredge 1985).

The development of gonads in saucer scallops starts when water temperatures have reached their peak (around 28 to 29°C) and begin to fall (Dredge 1981). However, Joll and Caputi (1995) reported no relationship between temperature and spawning for saucer scallops in Western Australia. Maximum gonad development occurred when water temperatures were near minimum (Dredge 1981). The spawning of saucer scallop can also be heat-manipulated in laboratory situations (Rose et al. 1988).. Dredge (1981) found that the fecundity and peak spawning of saucer scallops varied over relatively small distances, although *Amusium balloti* releases a lower number of eggs compared to other species of scallop such as *Pecten* (Rose et al. 1988).
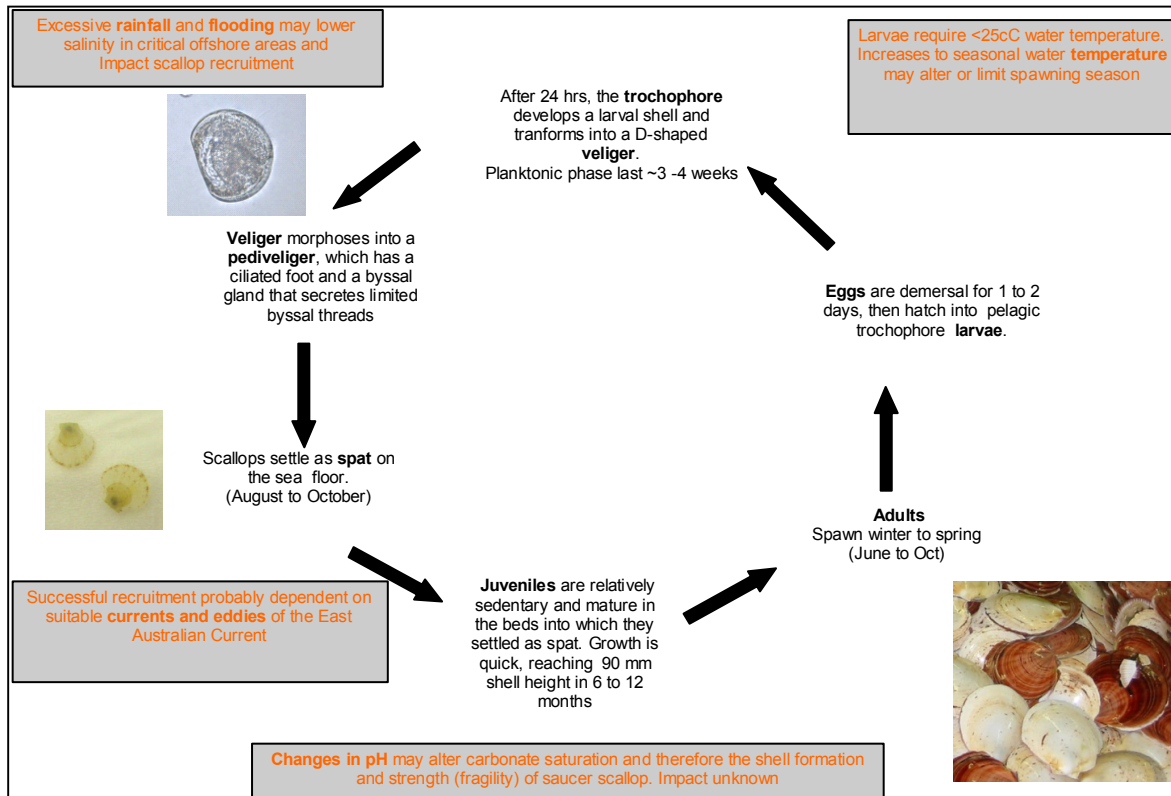
Figure 6.2. Generalised life cycle of saucer scallop (Images sourced from QDAFF and Sizhong Wang, QDAFF)

Saucer scallops have fast growth rates that vary with region and year. Water temperature is thought to be one of the significant drivers of variable growth in bivalves (Campbell et al. 2010). As water temperature can be a function of depth (i.e., warmer in shallower waters), saucer scallops in shallow waters (30 to 40 m) have faster reported growth rates than those from deeper waters (40 to 42 m) (Campbell et al. 2010). Water flow velocities that occur as a consequence of currents can affect food availability and feeding and thus growth. Campbell *et al*. (2010) reported slower growth rates than that reported by Williams and Dredge (1981) and suggested that the difference might be due to changed climatic conditions, competition for food or other density related processes or the season(s) during which the most recent work was conducted.

Reduced growth rate of saucer scallops was observed in Shark Bay scallops in November 2010 and February 2011 during the annual WA Fisheries fishery independent scallop survey (Pearce et al. 2011). Reduced growth was speculated to be linked to the higher water temperatures in Shark Bay (i.e., the WA marine heat wave) and possible variation in food availability. An additional (later than normal) settlement of scallops was also observed. The marine heat wave in WA, occurring between November 2010 and March 2011, from monthly satellite-derived sea surface temperatures (SST's) showed ocean warming of >2°C above average, with small areas (including Shark Bay) where temperatures were >3°C above average. Higher temperatures coincided with flood events in Shark Bay in December 2010 and February 2011, resulting in lowered salinity and higher turbidity in the waters of Shark Bay.

Distribution, habitat and environmental preferences

Saucer scallops are predominately a sub-tropical species that occurs in waters between 15°S and 25°S on the east coast of Australia and between 18°S and 35°S on the west coast of Australia (Dredge 2006). They are found in oceanic waters between 15 and 50 m deep, and in Queensland are most abundant in water depths >40 m and south of 20°S. Saucer scallops bury into sediment and as such occur in bare, sandy, rubbly or sponge garden habitats that have a soft but not muddy, sediment.
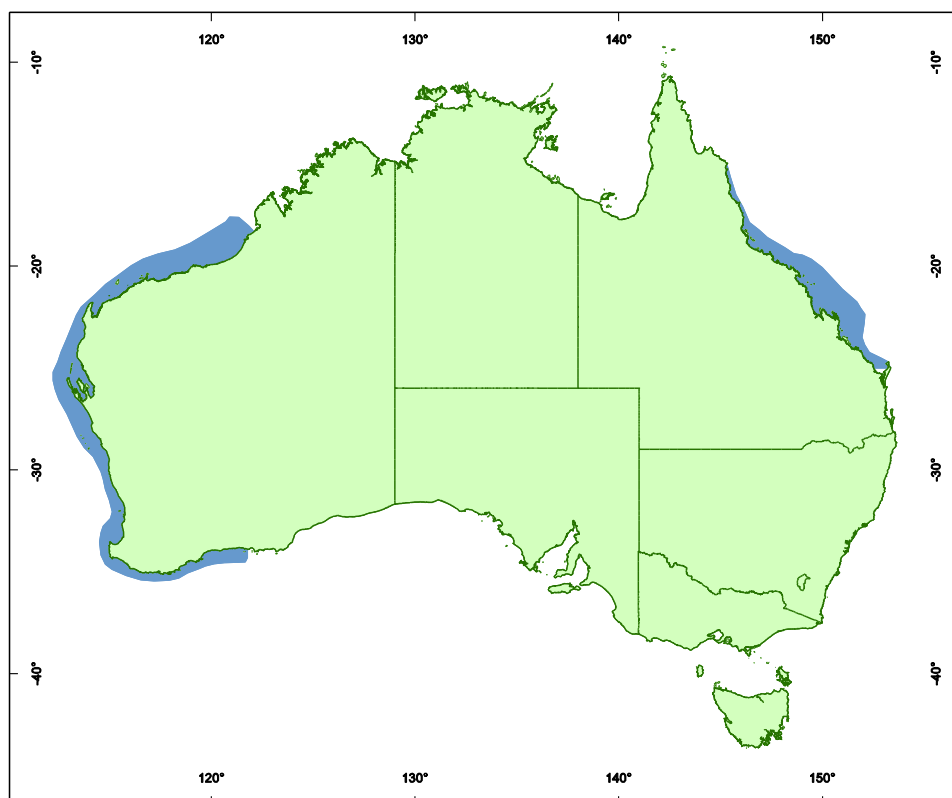
Figure 6.3. Australian distribution of saucer scallop.

Predators and prey

Saucer scallops are filter feeders and consume a variety of microscopic organisms in the wild, probably including phytoplankton and benthic diatoms. In aquaculture studies, saucer scallops have been fed algal cultures including *Tetraselmis suecia*, *Chaetoceros gracilis*, *C. clacitrans*, *Pavlova lutheri* and Tahitian *Isochrysis* (aff.) *galbana* (Cropp 1993).

Predators of saucer scallops include the slipper lobster (*Thenus orientalis*), the blue swimmer crab (*Portunus pelagicus*), the coral crab (*Charybdis cruciata*), loggerhead turtles (*Caretta caretta*), octopus and snapper (*Pagrus auratus*) (Dredge 2006).

Recruitment

Recruitment of saucer scallops is highly variable between years and at small spatial scales (Joll 1994; Joll and Caputi 1995). High recruitment into the Shark Bay scallop fishery is usually associated with

low mean sea levels (= weak Leeuwin Current) over the winter (spawning) months (Joll 1994). Joll and Caputi (1995) speculated that short-term variations in the structure and strength of the Leeuwin Current along the Western Australian coast provides periods of favourable and unfavourable environments within Shark Bay that subsequently determine recruitment success i.e. hydrological flushing of larvae (Caputi et al. 1996). Caputi *et al.* (1996) alternatively suggested that scallop recruitment may be linked to the effect of warmer water on spawning or fertilisation events. Similar impacts of the East Australian Current on recruitment success of saucer scallops in central Queensland have been speculated but not investigated.

Annual landings from the fishery in Western Australia are highly correlated with recruitment levels (Mueller et al. 2012). In Shark Bay, Mueller *et al.* (2012) suggested that environmental factors, such as tides, currents, and winds during scallop spawning and recruitment would be important factors in scallop recruitment because of the low correspondence between residual stock distribution (after fishing) and recruit distribution. Scallops in Shark Bay were affected by a "marine heat wave", where it is speculated that lowered salinity and higher temperatures may have exceeded the tolerances of adult scallops (Mueller et al. 2012).

Current impacts of climate change

Key points:
- Changes in major Australian currents (such as the Leeuwin and East Australian Current) are likely to impact recruitment of saucer scallops.

Recruitment variability of saucer scallops has been linked to weak Leeuwin Current in Western Australia in El Nino years (Caputi et al. 2010). An increased frequency of El Nino events may contribute to more years of good saucer scallop recruitment and therefore, be of benefit to WA fisheries (Caputi et al. 2010).

For other species of scallop (*Pectin maximus*) in the northern hemisphere, warmer spring temperatures (and not oxygen or chlorophyll a) have had a positive effect on gonad development and it has been suggested that this leads to increased gamete production and subsequent recruitment and catch (Shepard et al. 2010). It has also been suggested that warmer water, in the absence of excess food would have a negative effect on growth and reproductive development (Pilditch and Grant 1999). Whether this applies to saucer scallops in sub-tropical waters in unknown.

Sensitivity to change

Key points:
- Scallops have an unknown sensitivity to environmental variability.

Saucer scallops have restricted latitudinal ranges on the east and west coast of Australia, which probably reflects specific habitat requirements (bottom type, water depth, temperature and food availability) as well as dispersal mechanisms for larvae and spat (i.e., eddies).

Resilience to change

> **Key points:**
> • The resilience to change of saucer scallops is unknown, and as such has been classified as having a high level of ecological vulnerability to climate change.

Morison and Pears (2012) completed an expert based vulnerability assessment of the Queensland East Coast Otter Trawl Fishery and concluded that saucer scallops had a high level of ecological vulnerability from changed rainfall patterns and increased tropical storm intensity that could result in flooding, increased nutrients, pollutants and sediments. These events could reduce salinity and reduce available habitat and result in recruitment failure.

Morison and Pears (2012) also found that saucer scallops had high ecological vulnerability to higher sea surface temperature and ocean acidification, possibly effecting spawning triggers, larval development and inducing fragile shells. They also concluded that saucer scallops had a high level of ecological vulnerability to altered ocean circulation as larval dispersal and spat recruitment is (probably) dependent on eddies of the East Australian (and Capricorn) Current.

Other

> **Key points:**
> • Research into the links between environmental drivers and saucer scallop productivity in Queensland is needed.

Ecosystem level interactions
The abundance of some scallop populations have been significantly affected by predator levels (Hart 2006). This may or may not be the case for saucer scallops.

Additional (multiple) stressors
There are no known additional stressors on saucer scallops.

Critical data gaps and level of uncertainty
Little or no work has occurred linking indices of scallop abundance in Queensland (either catch, CPUE or fishery-independent recruitment surveys) with environmental influences, despite the strong anecdotal and scientific speculation of environmental drivers on saucer scallops. Further research into the links between scallop productivity and environmental drivers should investigate sea surface temperature, sea surface salinity, sea level anomalies (which captures warm and cold core eddies), and currents (see http://www.bom.gov.au/oceanography/forecasts/forecast-help.shtml).

Also, research is needed into the response of saucer scallops to changed ocean acidity to determine if saucer scallops are at risk of developing fragile shells as the pH of ocean water changes.

Acknowledgements

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7. Sea cucumbers (beche de mer, trepang)

Author: David J. Welch

The fishery

> **Key points:**
> - Recently active fisheries have been in WA, the NT and Qld while the Torres Strait fishery is generally inactive.
> - Species targeted vary among jurisdictions. Key species are: sandfish (*Holothuria scabra*), redfish (*Actinopyga echinites*), burrowing blackfish (*Actinopyga spinea*) and white teatfish (*Holothuria fuscogilva*).
> - Sea cucumber species are generally susceptible to overfishing and currently fishing for some species is banned due to overfishing.

Fisheries catch and status

The beche de mer or trepang fisheries across tropical and sub-tropical northern Australia are all hand collection fisheries by divers on snorkel, SCUBA or hookah, or by wading in shallow waters. These fisheries are almost exclusively commercial and are multi-species with species composition differing among jurisdictions. They are harvested primarily for the dried body wall and most are exported to Asia. In Western Australia (WA) only two species are taken (redfish – *Actinopyga echinites* and sandfish – *Holothuria scabra*) and in the Northern Territory (NT) the predominant species is *H. scabra* (Department of Fisheries, 2011; Northern Territory Government, 2011). On the Queensland (Qld) east coast there are at least 10 species taken and, although the species composition is variable through time, is dominated by two major species (burrowing blackfish – *A. spinea* and white teatfish – *H. fuscogilva*) (Anon, 2011a). In the Torres Strait historically there have been at least 16 different species targeted however in recent years there has been very little activity in the fishery due to the ban on taking some of the highest value species due to declining abundance. The prohibited species are *H. scabra* (since 1998), *H. whitmaei* and *A. mauritiana* (since 2003) (Anon, 2011b). Since 2008 there have been only 3 active fishers in the Torres Strait (all Traditional Inhabitant licensees) and very little commercial catch has been reported since 2005. Because there are only 3 active fishers catch figures in recent years have not been publicly available due to confidentiality reasons (Anon, 2011b). Prior to 2007 the WA fishery was essentially a single species fishery with ~99 % of the catch being *H. scabra*. In the past two years in Qld the fishery is targeting curryfish (*Stichopus hermanni*), mainly due to its higher value and improvements in processing.

Total commercial catch was 121 t in WA in 2009-10 (redfish – 71 %, sandfish – 29%), 22 t in the NT in 2010[1], and 352 t in Qld in 2009-10 (burrowing blackfish – 70 %, white teatfish – 20 %, curryfish – 9 %). In WA sandfish has been the major species historically and targeting of redfish has only picked up

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[1] The average annual catch over the period 1998 to 2007 was 235 t and low catches in the past three years can be attributed to the sole NT licensee shifting most of his effort to WA during those years.

since 2007 (Figure 7.1). In the NT catch of sandfish has been highly variable with effort (and catch) noticeably decreasing since 2002 (Figure 7.2). Currently sea cucumber fisheries are considered to be fished at sustainable levels in WA and Queensland (Anon, 2011a; Department of Fisheries, 2011) while in the NT they have not been assessed (Northern Territory Government, 2011). On the Great Barrier Reef black teatfish (*H. whitmaei*) is over-fished and the fishery for them has been closed since 1999 (Uthicke et al, 2004). Between 2001-02 and 2009-10 sea cucumber catch on the GBR has been fairly stable (Figure 7.3). The catch value of the WA fishery was estimated to be $330,000 for the year 2010 (Department of Fisheries, 2011), and the catch from the Qld fishery was estimated to value $4.9 M in 2009-10 (Anon, 2011a).
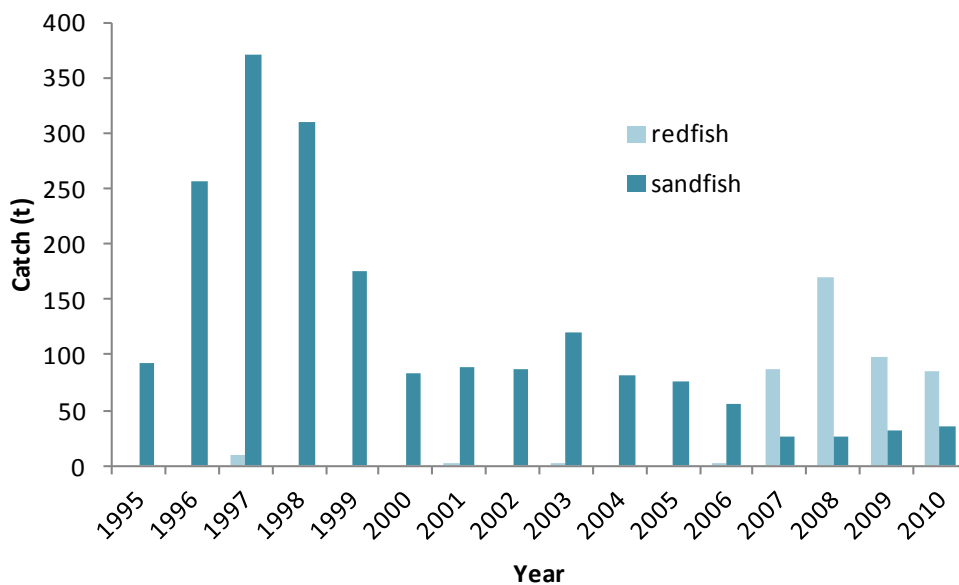
Figure 7.1. Catch of the two major sea cucumber species in commercial fisheries in WA from 1995-2010 (Source: Department of Fisheries, 2011).

Fisheries Management
Although fisheries for holothurians (sea cucumbers) globally have a history dating back hundreds of years, growing demand and resulting high value in SE Asia meant fisheries targeting increased during the 1980s. Throughout their history sea cucumber fisheries have been characterised by boom-and-bust cycles due to life history and ecological characteristics of holothurians. Management of wild sea cucumber fisheries therefore require careful and conservative strategies to ensure sustainability (Anderson et al, 2011; Purcell et al. 2011).
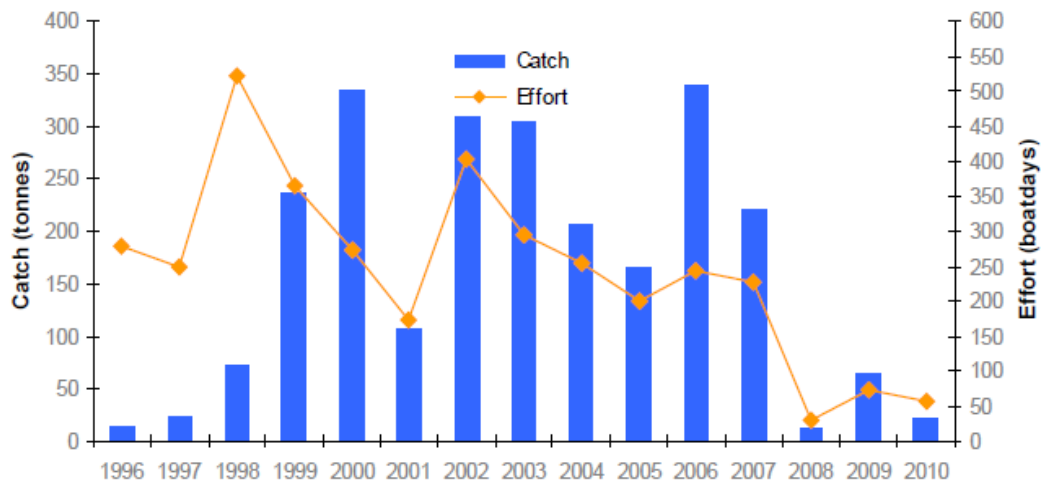
Figure 7.2. Catch of sandfish by NT commercial fishers during 1996-2010. (Source: Northern Territory Fisheries, 2011).
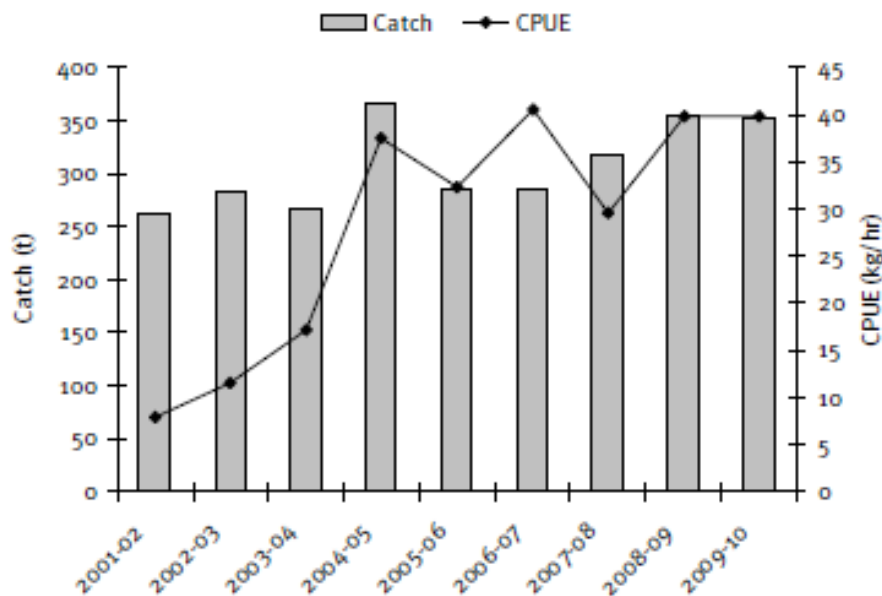
Figure 7.3. Total catch of sea cucumber on the GBR from 2001-02 to 2009-10. (Source: DEEDI, 2011).

Western Australia

The WA fishery operates from Exmouth Gulf to the NT border and is managed through a number of input controls. These include limited entry, a maximum number of divers, hand collection only, size limits depending on the species and gear restrictions. There are 6 vessels licensed to fish however in the past three seasons only two of these have been operating in the fishery. Catch is reported separately for the two species, as is effort, which is spatially separate due to the different habitats the two species occupy. This also means that management performance measures are species-specific (Department of Fisheries, 2011).

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Northern Territory

The NT fishery covers the entire coastline to 3 nautical miles but the major areas where catch and effort is focused are the Cobourg Peninsula and Groote Eylandt along the Arnhem Land coast. Key management strategies used include limited entry (6 licenses of which all are held by the one licensee), separate management zones with limited entry by licence, a limited number of collectors, and hand collection only (Handley, 2010).

Queensland

The fishery in Queensland can operate from Tin Can Bay to Cape York however most of the historical effort has been in north Qld reef areas north of Townsville. Working depths are greater than in other jurisdictions (30 m) due to the targeting primarily of white teatfish. Management controls include a total allowable catch of 361 t of which there are individual species quotas for black teatfish (currently not fished and has 0 t quota), white teatfish, and other species, limited entry (currently 18 transferable licences of which only 7 reported catch in 2009-10), minimum size limits for key species, gear limitations, area closures through the Great Barrier Reef zoning, and a rotational zoning scheme (RZS). The RZS is an industry led scheme that divides the total fishing area into 154 fishing zones which can be fished only one in every three years and even then for a maximum of 15 days annually. Catch (but not effort) is reported by species and there are performance measures for the key species (Anon, 2011a).

Life history

> **Key points:**
> - Sea cucumbers generally have very low replenishment rates making them susceptible to overfishing and slow at recovering from perturbations.
> - Successful recruitment is likely to be influenced (in part) by adequate densities of spawning adults.
> - Sea grass appears to be an important habitat for *H. scabra* recruits.

Life history information documented here is generalised for all sea cucumbers unless individual species are indicated. There is a general focus on *H. scabra* due to its relative fishery importance and high level of knowledge relative to other species. It is acknowledged however that the species-specific biology and ecology of sea cucumbers can be highly variable (Conand, 2006).

Life cycle, age and growth

Tropical and sub-tropical sea cucumbers are primarily broadcast spawners with fertilisation taking place in the water column however some species exhibit asexual reproduction by transverse fission (Uthicke, 2001a). Broadcast spawning is generally annual or bi-annual, though some species (eg. *H. scabra*) are capable of spawning year round in warmer equatorial waters (Morgan, 2000a), while asexual reproduction occurs in early winter (Uthicke, 2001a). Temperature appears to be the main cue to spawning though there may be other exogenous cues and is often linked to lunar cycles. Spawning seasons are generally Spring-Summer and can vary by species and spatially (Morgan, 2000a) with a few species preferring to spawn during winter (eg. black teatfish, *H. whitmaei*; Shiell and Uthicke, 2006). *H. scabra* is currently the only tropical sea cucumber that can be mass reared in

hatcheries, although recent developments suggest that broad-scale culture of curryfish (*Stichopus* spp.) may also be possible (Hamel et al, 2001; Hu et al, 2010). Holothurian broodstock are induced to spawn generally by raising the tank water by 3-5 °C (Morgan, 2000b; D. Welch, pers. obs.), however the introduction of micro-algae into tank water has also been shown to trigger spawning in a number of holothurians species (Battaglene, 1999).

Egg development in holothurians is generally short (24 hrs) and the planktonic larval duration varies among species and for some of the key harvested species ranges from 12 - 22 days (Ramofafia et al, 2003). In cultured situations for *H. scabra* temperatures are kept between 26 °C and 29 °C during larval development. Larvae feed on different species of micro-algae and successful metamorphosis has been shown to be dependent on the algal species consumed. One of the better algal species, *Chaetoceros muelleri*, is very tolerant of high temperatures (Battaglene, 1999). Larvae develop through the feeding stage auricularia, the non-feeding doliolaria, and the pentactula stage that develops tentacles and settles (Fig 1.; Ramofafia et al, 2003). Preferred habitat types for settlement appear to be on sea grass leaves for *H. scabra* (Mercier et al, 2000a) with cues including the presence of particular food types such as diatoms and certain bacteria (Battaglene, 1999). Generally, however, very little is known of the larval movement and settlement processes in the wild (Conand, 2006).

Growth rates of sea cucumbers are also poorly understood but are generally believed to be slow with low overall productivity (Uthicke et al, 2004). Hu et al (2010) were able to grow curryfish (*Stichopus* sp.) juveniles to approximately 20 cm within 7 months in a hatchery. Aging of holothurians in the wild has not been possible however modelling by Uthicke et al (2004) suggested that *H. whitmaei* are long-lived (potentially several decades).

Distribution, habitat and environmental preferences
Sea cucumbers are benthic animals found mostly on soft substrates such as sand and mud however they are usually associated with sea grass, algae and corals.
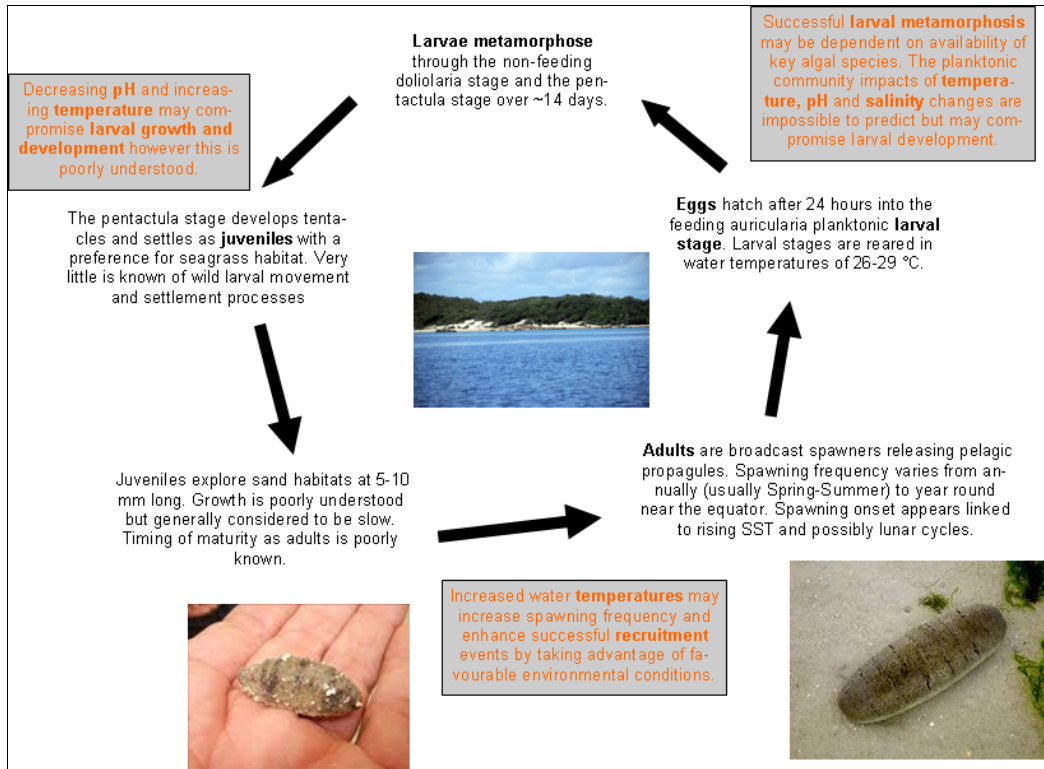
Figure 7.4. Life cycle of the sandfish, *Holothuria scabra***, in cultured systems where larvae are induced to settle on plates. This cycle is typical of many tropical sea cucumbers (Source: Battaglene, 1999). Images: GBRMPA.**
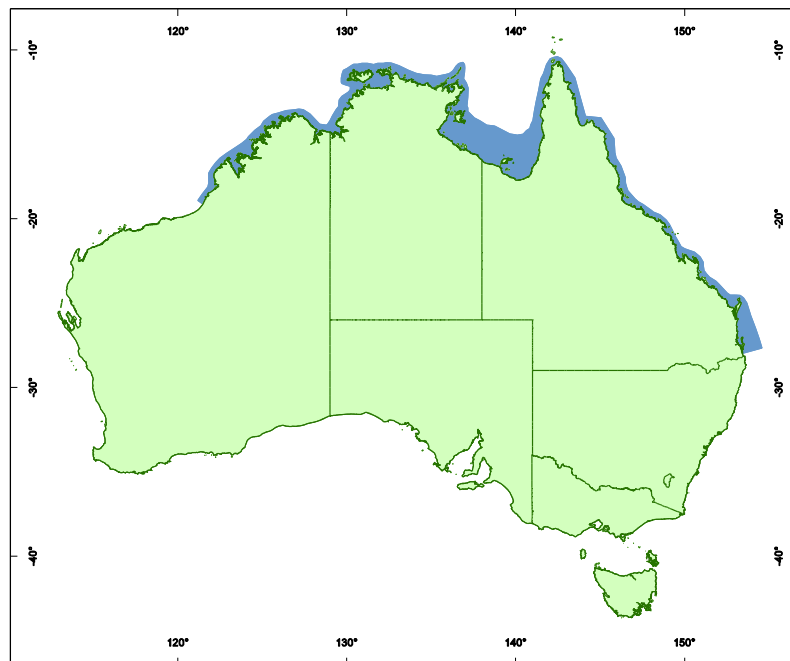
Figure 7.5. Australian distribution of *H. scabra***.**

Predators and prey

Sea cucumbers are benthic deposit feeders feeding on micro-algae, bacteria, diatoms and detritus. Ecologically they play a very important role in bioturbation of the upper sediment layers and provide an important nutrient recycling function thereby increasing the benthic productivity of oligotrophic systems such as coral reefs (Uthicke et al, 2004). Predation is possibly greatest during the larval stage when other larvae can eat them.

Recruitment

Modelling by Uthicke et al (2004) suggested that *H. whitmaei* recruitment is low and sporadic due to the apparent slow rate of population recovery after overfishing. A study of *H. scabra* in the Solomon Islands found monthly recruitment of newly-settled juveniles (Mercier et al, 2000b). Typically, successful recruitment of low mobility marine organisms such as sea cucumbers require adequate adult densities to ensure successful fertilisation of released eggs.

Current impacts of climate change

There are no known documented current impacts of climate change.

Sensitivity to change

> **Key points:**
> - Studies of environmental sensitivity for key Australian fishery holothurian species are generally absent from the literature.
> - From existing studies, changes in temperature, salinity and pH may affect holothurian distribution and abundance with early life history stages being most vulnerable.

Rearing of sea cucumbers for restocking and/or to supplement over-fished wild stocks has elucidated optimal conditions for rearing larvae of some tropical sea cucumber species and gives some insight into preferences and tolerance limits. These include *H. scabra*, *Actinopyga echinites* and *H. atra* and optimum temperature ranges for these species was between 27 and 30 °C (Battaglene, 1999; Chen and Chian, 1990; Ramofafia et al, 1995). In a study of a tropical sea cucumber commercially harvested in India until the fishery collapsed in 2001, the effects of temperature, pH and salinity on growth and survival of *H. spinifera* larvae were experimentally determined (Asha and Muthiah, 2005). When comparing the temperatures 20, 25, 28 and 32 °C they found that growth and survival was far greatest at 32 °C, however this also reduced the time to settlement and therefore is likely to reduce dispersal capabilities. Growth was significantly affected by salinity with 35ppt the best salinity when compared with 15, 20, 25, 30 and 40ppt. Comparing pH of 6.5, 7.0, 7.5, 7.8, 8.0, 8.5 and 9.0 they found that survival was significantly enhanced at 7.8 (83 % survival), at 9.0 there was 0% survival, and at all other pH regimes survival ranged between 47 and 60 %. Comparisons of growth however, could not be carried out since deformities occurred in larvae in all pH regimes except for 7.8 (Asha and Muthiah, 2005).

The major holothurians targeted by tropical fisheries possess microscopic components in their body wall called spicules which for the internal skeleton. These spicules are calcareous as is the peripharyngeal ring (Conand, 2006). The effects that ocean acidification may have on these species is unknown. Experimental studies on different species of the Phylum Echinodermata, of which sea

cucumbers are part of, to changes in seawater pH had varying results and included reduced fertilisation rates and reduced larval sizes (sea urchin, *Echinometra mathaei*, Kurihara and Shirayama, 2004) and reduced survival and larval size (sea urchin, *Tripneustes gratilla*, Clark et al, 2009). Given species-specific responses to changes in pH, potential impacts on sea cucumbers will remain highly uncertain without studies on the species of interest.

Resilience to change

Key points:
- Due to low mobility the capacity for sea cucumber species to move away from unsuitable environmental conditions is poor.
- The timing of spawning of most tropical species means they are likely to be resilient to increases in temperature. The notable exception is the black teatfish that is a winter spawner in the tropics.

Population genetic techniques showed that populations of *H. whitmaei (*previously *H. nobilis)* on individual reefs in the GBR are highly connected and that even populations from West Australia and on reefs in the Coral Sea are potential sources of recruits (Uthicke & Benzie, 2000*b*, 2003). This is not likely to reflect contemporary scales given the low mobility of adults and the relatively short larval duration. Conversely, studies have found that gene flow in *H. scabra* in New Caledonia, north-eastern Australia and the Solomon Islands was restricted even on small spatial scales (Uthicke and Benzie, 2001; Uthicke and Purcell, 2004). Stock structure is not well understood for other northern Australian species and based on the above studies is likely to vary depending on the species.

Holothurians feed on micro-organisms in benthic substrates. Micro-organisms form the basis of food webs and, although holothurians are dependent on this food source, their availability is not likely to be limiting. Spawning seasons of many sea cucumbers are during Spring-Summer and with forecast temperature increases this may begin earlier or spawning may even become year round as seen in *H. scabra* close to the equator (Morgan, 2000a). Reproductive success in species that spawn during winter (eg. *H. whitmaei*) may be compromised and any such impacts will be evident in more northern tropical regions first. Upper thermal limits for spawning and larval growth and development are not known however cultured holothurian larval stages are currently raised in 26 – 29 °C water (Battaglene, 1999), and some species larval survival and growth is better at higher temperatures (32 °C) (*H. spinifera*, Asha and Muthiah, 2005).

Other

Key points:
- Sea cucumbers play an important ecological role in maintaining benthic productivity by remineralising organic nutrients. They also play an important role in buffering ocean acidification at local scales.
- Specific studies on key commercial sea cucumber species are needed to assess the effects of altered environmental conditions, particularly during the early life history stages.

Ecosystem level interactions

Holothurians are known to play an important ecological function. For example, on coral reefs holothurians are able to bioturbate the upper 5 mm of sediment (equivalent to 4.6 t/ha) annually (Uthicke, 1999). Holothurians feed on bacteria, diatoms, and detritus (Yingst, 1976; Moriarty, 1982) and by digesting these organisms they remineralize large quantities of organic nutrients (Uthicke, 2001b). This important nutrient recycling loop increases the benthic productivity of oligotrophic systems such as coral reefs (Uthicke & Klumpp 1998; Uthicke 2001c). Therefore, it is possible that impacts that decrease holothurian populations will result in reduced overall productivity of coral reefs.

Perhaps more importantly in the context of climate change, and in particular to the forecast acidification of sea water with increased atmospheric CO_2, is the role that holothurians play in the dissolution of $CaCO_3$ on coral reefs. Schneider et al (2011) examined two commercially exploited sea cucumber species (*S. herrmanni* and *H. leucospilota*) at One Tree Island on the southern Great Barrier Reef and determined that, as well as being important in the natural turnover of $CaCO_3$, their role in the dissolution of CaCO3 sediment was also an important source of alkalinity. Sea cucumber therefore may play a role in buffering ocean acidification at least at local scales on coral reefs, thereby reducing associated impacts such as reduced coral growth and larval survival.

Additional (multiple) stressors

Holothurian fisheries have a history of being 'fished down' and have followed a cycle of periods of fishing and recovery. Most species therefore appear to be prone to overfishing. The effect of poor water quality on nearshore species is poorly understood.

Critical data gaps and level of uncertainty

Estimates of age and growth rates of holothurians are rare and subject to considerable error. The small larvae produced from external fertilization of gametes cannot be physically tagged, and newly settled animals are usually rarely detected, leading to a major gap in knowledge concerning the sources and numbers of recruits. The sensitivity of important fishery holothurian species to changes in the environment are unknown.

Acknowledgements

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8. Tropical rock lobster, *Panulirus ornatus* (ornate rock lobster)

Authors: David J. Welch and Julie Robins
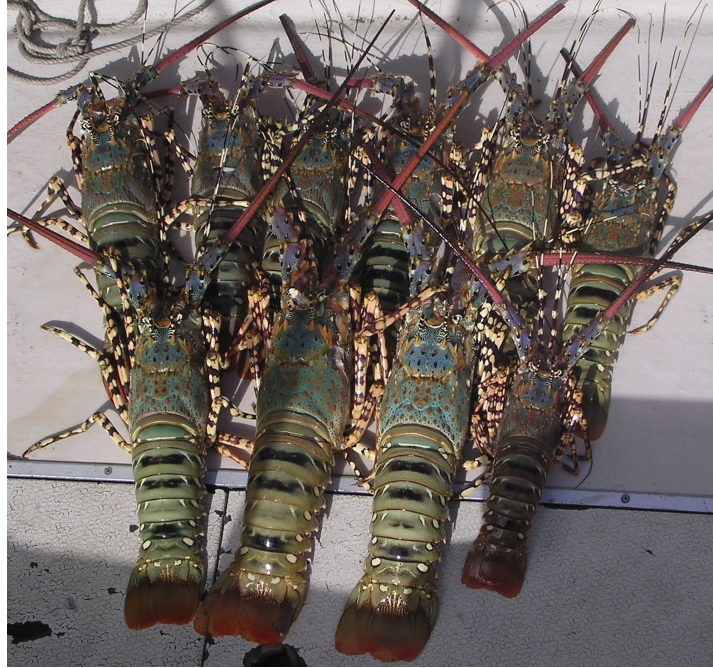
Recreationally caught tropical rock lobster. Photo: D. Welch.

The fishery

> **Key points:**
> - Only Queensland and Torres Strait support significant fisheries in Australia.
> - Both of these fisheries are relatively low yield but highly valuable.
> - Catch levels are thought to be determined by variable annual recruitment.

Operational characteristics

Fisheries for tropical rock lobster in northern Australia only occur in NE Queensland and Torres Strait waters as separate fisheries based on jurisdiction (State and Commonwealth). The key specie targeted is the ornate rock lobster, *Panulirus ornatus*, and although other species are taken they make up insignificant portions of the catch (eg. < 2% of the Qld fishery). The fisheries are dive-based with commercial collection methods by divers using hookah or freediving and collecting by hand, nooses (snares), or in some cases using hand spears. Divers work around coral reefs in depths up to 20 m and operate almost exclusively during daylight hours (AFMA, 2010; DEEDI, 2010). There is a small domestic market for product however most are exported overseas to mainland China via Hong Kong (Pitcher et al., 2005). The major product form is as frozen tails however there is a live component also (AFMA, 2010; DEEDI, 2010).

Torres Strait fishery

Characteristics

The area of the fishery is in Torres Strait from the tip of Cape York to the northern border of the Torres Strait Protected Zone. The commercial fishery within this zone is shared between Australia and Papua New Guinea under a formal arrangement (AFMA, 2010). There is a small recreational fishery within the Torres Strait. The Torres Strait commercial fishery is comprised of two sectors – the non-Indigenous (TVH) sector and the Traditional Inhabitant (TIB) sector (AFMA, 2010). Dive operations consist either of a mother vessel from which a number of smaller (4-6 m) tender vessels operate with divers working from each tender (TVH operators), or of a small 4-6 m vessel with divers using solely freediving (TIB). There are 13 TVH primary licences with 34 tenders attached to these, while in the TIB sector there are currently 470 licenses of which only 293 are active (as of September, 2010).

Fisheries catch and status

The fishery catch is managed through a quota system with an annual Total Allowable Catch (TAC) that is shared between Australia and Papua New Guinea. The historical catch from the fishery is variable from year-to-year and is thought to be driven by variable recruitment. In 2009 the total catch from the fishery was valued at $AU7.5 M, and was comprised of 228 t (live weight) for the Australian portion (Figure 8.1) and 114 t for the PNG portion. For the 1989 to 2009 time period Papua New Guinea fishers took approximately 31% (range: 19 – 57%) of the total Torres Strait catch. Within the Australian catch, historically the TVH sector has taken the most however in recent years, due to effort controls (regulated and voluntary), most of the catch is taken by the TIB sector and in 2009 they took 59% of the catch (Table 8.1) (AFMA, 2010). The most recent assessment of the fishery is that it is not overfished nor is it subject to overfishing.
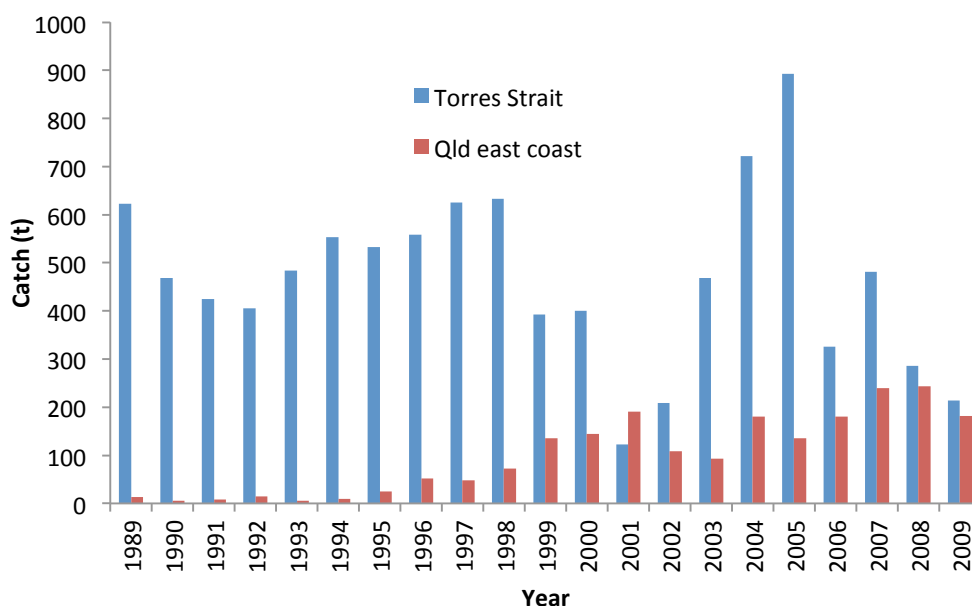
Figure 8.1. Historical Australian commercial catch for the Torres Strait and commercial catch for the east coast of Australia for the years 1989 to 2009. Catch has been converted to tonnes live weight (Source: AFMA, 2010).

Fisheries Management

Management is under the Torres Strait Fisheries Act 1984 and through policies agreed to under the protected Zone Joint Management Authority. Regulations include restrictions on the number of TVH licenses and how many tenders per primary vessel, however there is no limit on the number of TIB licenses that can be issued. Other regulations include taking of lobster only by hand or hand-held implements, a ban on the use of hookah during December and January each year, a minimum tail size of 115 mm or a minimum carapace length of 90 mm, and bag limits of 3 per person, or 6 per dinghy for recreational fishers and traditional fishing.

Table 8.1. Catch (whole weight in tonnes) of the non-Indigenous (TVH) sector and the Traditional Inhabitant (TIB) sectors of the Torres Strait Rock Lobster fishery from 2001 to 2009 (Source: AFMA, 2010).

Year	TVH	TIB	Total	TIB (%)
2001	70	53	123	43
2002	144	65	209	31
2003	350	118	468	25
2004	465	257	722	36
2005	523	370	893	41
2006	130	196	326	60
2007	257	238	495	48
2008	98	177	274	65
2009	88	126	214	59

Queensland fishery

Characteristics

The fishery comprises commercial, recreational and Indigenous sectors. The commercial sector is restricted to the far northern region of the Great Barrier Reef (~ north of Princess Charlotte Bay) and the Gulf of Carpentaria. Dive operations consist of a mother vessel from which a number of smaller (4-6 m) tender vessels operate with divers working from each tender. There are 28 primary licences with 93 tender licences attached to these, however only 11 primary licences accessed the fishery in 2009. There is also 5 t catch limit allowed annually for Indigenous Fishing Permit holders. Catch from the Gulf of Carpentaria is negligible. Recreational catch is taken south of the commercial fishery area on the east coast and extends south to at least the Qld border.

Fisheries catch and status

The commercial catch has been slowly but steadily increasing since 1995 from 25 t to 192 t in 2009 (Figure 8.1). From surveys conducted in 2001 and 2005 it was estimated that the Indigenous sector took 13,000 lobsters, while recreational fishers were estimated to take 17,000 lobsters (Henry and Lyle, 2003; DEEDI, 2010). Although the fishery is considered fully exploited, a recent stock assessment concluded that current catch levels (now regulated by a TAC) is within MSY estimates,

and the fishery is considered as "being managed in a precautionary and sustainable manner" (DEEDI, 2010).

Fisheries Management

The Queensland east coast fishery and the Torres Strait fishery have been shown to comprise the same lobster stock (Pitcher et al, 2005). As such, management of each fishery has been moving more towards being complementary. Management of the Qld fishery is the responsibility of Fisheries Queensland, part of DAFF. Management of the fishery is by limited commercial entry, a Total Allowable Commercial Catch system, mated and egg-bearing females cannot be taken by commercial fishers, a seasonal closure between October 1 and January 31 within the commercial fishing area, minimum size limits consistent with the TS fishery, and recreational bag limits (DEEDI, 2010).

Life history

> **Key points:**
> - In NE Australian waters many *P. ornatus* adults undergo an annual migration of between 70 and 500 km into deep continental shelf waters of the Coral Sea for spawning.
> - Larvae drift in oceanic waters of the NW Coral Sea for approxmately 6 months prior to settlement.
> - NW Coral Sea currents are highly important for recruitment dynamics in NE Australia and Torres Strait.

Life cycle, age and growth

The breeding season for adults is between November and April. Adult breeding *P. ornatus* migrate to breed at 2.5 to 3 years old and females outnumber males in the breeding migrations 2:1. Large males and one-year-old lobsters do not migrate. Shorter migrations are undertaken by lobster on the north-east coast of Queensland (average 70km), whilst larger migrations are undertaken by lobsters in Torres Strait (up to 511km). Breeding sites include deep water (40 to 120m) areas on the continental shelf outside the Great Barrier Reef and Yule Island in the Gulf of Papua. Breeding sites on the Great Barrier Reef are predominantly in the far north however breeding sites are known to occur south to at least Townsville (19° S) (Bell et al, 1987). Some adults migrate from reefs in Torres Strait from August to November. Lobsters that migrate to Yule Island generally do not survive after breeding (Pitcher et al, 2005). *P. ornatus* are highly fecund and multiple broods may be carried and reared during one spawning season, although the first brood is thought to represent the major spawning within a season. In captivity females produce an average of 3 batches each breeding season at 28 °C (M. Kenway, pers. comm.). Queensland and Torres Strait *P. ornatus* are considered to be a single genetic stock with Torres Strait and far NE areas being source populations to areas of the GBR further south (Pitcher et al, 2005).

Eggs are fertilised as they exit the female's body and attach to the pleopods, where they are carried for approximately 35 days at 29 °C (Pitcher et al, 2005). Under captive conditions in tanks at 28 °C females carry eggs for 26 days (M. Kenway, unpublished data). Larvae hatch as phyllosoma that are carried by wind and tides in the plankton of oceanic waters of the NW Coral Sea and go through as many as 24 morphological stages over approximately 6 months (Pitcher et al, 2005; Smith et al, 2009). The larvae develop into the peurulus stage that is an active non-feeding swimming stage that

seeks out suitable benthic habitat. The peurulus swims across the continental shelf to settle in coastal areas as benthic juveniles. Sub-adult lobsters (~95 mm CL) move offshore during March/April to mid-shelf reefs. In the Torres Strait sub-adults move widely throughout the region seeking suitable reef habitat and/or large beds of bastard shell, *Pinctada albina* (M. Kenway, pers. comm.).

Growth of *P. ornatus* has been generalised using the von Bertalanffy growth function and was derived from tag-recapture and aquarium data. Longevity is estimated to be approximately 8 years at which *P. ornatus* have a carapace length of approximately 150 mm (Phillips et al, 1992; Skewes et al, 1997). In wild populations larger individuals tend to be males, possible due to higher natural mortality rates on females from the annual breeding migration and egg brooding.
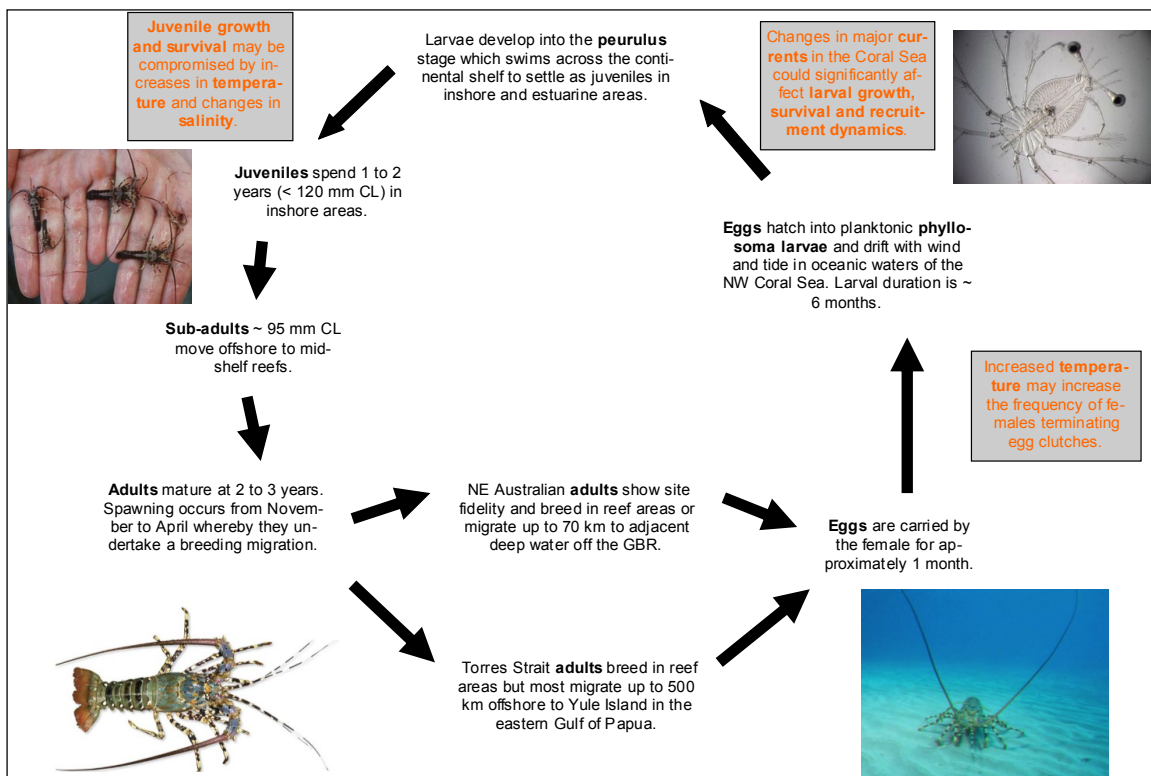
Figure 8.2. Generalised life cycle of the ornate rock lobster, *Panulirus ornatus*, from the NE region of Australia and the stages of potential environmental driver impacts. Images: Queensland DEEDI; Pitcher et al, 2005.

Distribution, habitat and environmental preferences

Tropical rock lobsters occur in northern Australia and inhabit reef tops, reef slopes and rocky inter-reef areas, up to 200m deep on the continental shelf (Kailola et al, 1993). *P. ornatus* are known to have a broad habitat use including deep (> 200 m) oceanic waters to muddy reefal areas adjacent to estuaries and river mouths, which reflects a very wide distribution (Pitcher et al, 2005). They prefer reef habitat and within NE Australia and Torres Strait can be found across the entire continental shelf.
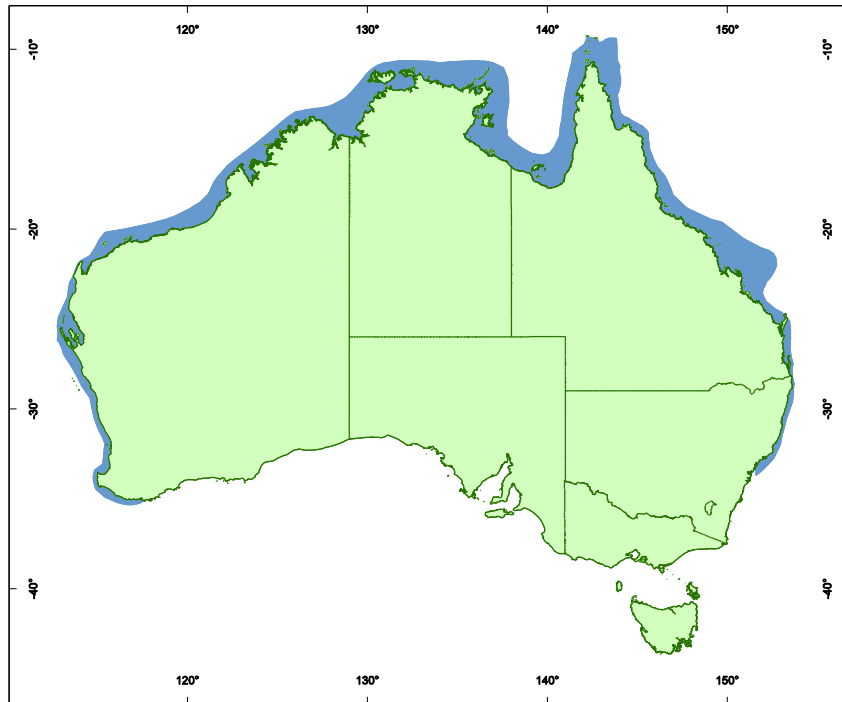
Figure 8.3. Distribution map for *P. ornatus* within the Australian region.

Predators and prey

P. ornatus have been described as opportunistic carnivores that feed mainly on benthic invertebrates. Several studies that have analysed gut contents of juveniles and adults have found a variety of different molluscs including bivalves, chitons and gastropods, other crustaceans such as barnacles, crabs and other decapods, polychaete worms and echinoderms (see Williams, 2007). Commercial divers in the Torres Straits and on the east coast maintain that they target *P. albino* beds when collecting rock lobsters suggesting this mollusc species is important habitat and/or as a prey item (M. Kenway, pers. comm.).

Several early studies have documented the capability of planktonic phyllosoma stage larvae of various other lobster species to feed on a variety of different planktonic prey items. These include eel and fish larvae, trochophore veliger larvae, calanoid copepods, hydromedusa, polychaetes, ascidian larvae, crab zoeas, chaetognaths and salps (eg. see Batham, 1967). Given the similarity in development among species it is assumed that *P. ornatus* larvae possess similar feeding capabilities. Some very early studies documented the attachment of phyllosoma larvae to medusa (Thomas, 1963; Hernnkind et al, 1976) however to this day it is still unclear whether this is a feeding mechanism or something else (eg. predator avoidance). There are no published studies on the natural predators of *P. ornatus*.

Recruitment

In the NE Australian region the distribution of *P. ornatus* phyllosomas and pueruli in relation to ocean currents support the hypothesis that phyllosomas are transported from the Gulf of Papua breeding grounds by the Hiri boundary current into the Coral Sea Gyre and then by surface onshore

currents onto the Queensland coast, Torres Strait and SE Papua New Guinea. There appears to be distinct regions that act as recruitment 'sources' and 'sinks' which is determined by the bifurcation of the South Equatorial Current off the GBR approximately adjacent to Cooktown on the NE Queensland coast. Areas to the north of this bifurcation can be termed both source and sink regions and to the south as a sink region (Figure 8.4) (Dennis et al., 2001; Pitcher et al, 2005). The peak timing of settlement in NE Queensland occurs during winter (June-August) in most years however the seasonality of settlement is highly variable.
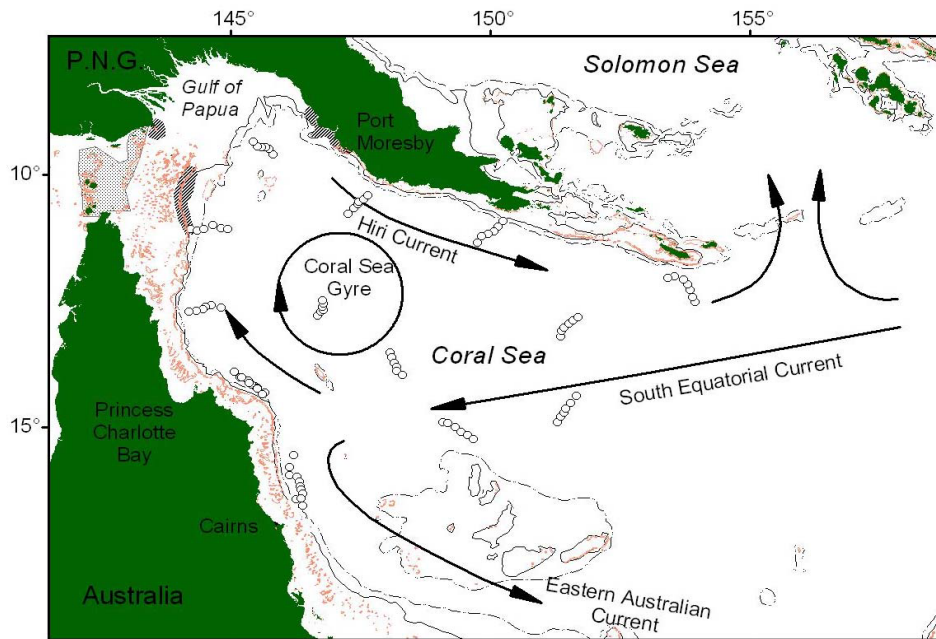
Figure 8.4. Map of the NW Coral Sea region showing the major currents that influence the eventual recruitment of *P. ornatus* larvae. Circles indicate areas of plankton sampling conducted during May 1997 and hatched areas indicate the known breeding grounds (from Dennis et al, 2001).

Current impacts of climate change

Key points:
- Catchability has been affected during recent warmer than average years when lobsters move to deeper waters less accessible to divers.

Commercial divers in Torres Strait reported that during recent "hot" years *P. ornatus* moved to deeper cooler water making them less accessible for capture (Welch and Johnson 2013).

Sensitivity to change

Key points:
- *P. ornatus* appear to have distinct temperature preferences at all life history stages and females even terminate egg clutches at $\geq 32\ °C$.
- *P. ornatus* juvenile growth and survival appear to be moderately influenced by temperature and salinity.

Temperature and salinity tolerances of *P. ornatus* were investigated by Jones (2009). He found that juvenile growth was significantly affected by temperature with maximum growth in 25 – 31 °C water and the optimal temperature being 27 °C. Salinity was also found to have a significant effect on juvenile growth and survival with lowest survival but fastest growth at 35 ppt. Sachlikidis et al (2010) found that *P. ornatus* terminated their egg clutches in temperatures ≥ 32 °C. Currents in the NW Coral Sea are extremely important for carrying *P. ornatus* larvae and the determination of areas of settlement (Pitcher et al, 2005).

Western rock lobster (*Panulirus cygnus*) are thought to have a decrease in their size at maturity due to rising sea temperatures. The Leeuwin Current (influenced by the Southern Oscillation Cycle) is also thought to influence puerulus settlement (Caputi et al, 2010).

Resilience to change

Key points:
- *P. ornatus* have broad habitat preferences and extensive available habitat on the east and west coasts.
- The future of the Torres Strait/Queensland fisheries may be dependent on large-scale changes to ocean currents in the NW Coral Sea.

P. ornatus have a broad geographical range and a broad habitat preference. Within the key fishery regions of northern Australia a single genetic stock is present with distinct 'source' and 'sink' regions (Pitcher et al, 2005). Although larval development is approximately 6 months, under culture situations there is evidence that this period can be as short as 4 months indicating some plasticity in their early development (Smith et al, 2009). Experimental studies have shown juvenile growth to be maximised between 25 – 31° C water temperatures (Jones (2009), while Sachlikidis et al (2010) found that *P. ornatus* terminated their egg clutches in temperatures ≥ 32° C, making them susceptible to projected SST increases. The long larval phase may be a significant limiting factor to successful recruitment depending on the nature of future change, particularly with respect to ocean currents in the NW Coral Sea.

Other

Key points:
- Better understanding of how ocean currents in the NW Coral Sea may change under climate change will allow more certain predictions of the recruitment dynamics of *P. ornatus* in the future.

Ecosystem level interactions

The role of *P. ornatus* larval stages play in the plankton in terms of predator-prey interactions with other plankton species is poorly understood and may be significant especially given their larval duration.

Additional (multiple) stressors

P. ornatus are commercially and recreationally fished in the Torres Strait and the NE GBR, however elsewhere they are only lightly harvested by recreational and Indigenous fishers. Juveniles use inshore and estuarine habitats during the first 18 months after settlement and so pollution and runoff may be additional stressors at various times and places, although currently most recruitment is in the far northern region of the GBR and Torres Strait where pollution and runoff impacts are relatively low compared with areas further south.

Critical data gaps and level of uncertainty

Variation in annual environmental conditions determines the successful recruitment of *P. ornatus* which drives the Torres Strait and Queensland fisheries. A critical knowledge gap therefore is the likely change in ocean currents in the NW Coral Sea and how these may influence the recruitment dynamics of *P. ornatus*.

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FINFISH AND SHARKS

9. Barramundi, *Lates calcarifer*

Authors: Emily Lawson, Thor Saunders and Julie Robins

A Northern Territory barramundi. (Image sourced from NT Fisheries).

Barramundi is an important and iconic species throughout northern Australia and are important for all fishing sectors economically, socially and culturally.

The fishery

> **Key points:**
> - Barramundi represent a very important fishery species across northern Australia for recreational, commercial and Indigenous sectors.
> - The annual harvest is regionally variable and is often positively linked to rainfall/riverflow.

Western Australia

In Western Australia barramundi are captured in the commercial Kimberley Gillnet and Barramundi Managed Fishery (KGBF) which operates in the nearshore and estuarine zones of the North Coast Bioregion from the border between Western Australia and the Northern Territory (~129° E) to the top of Eighty Mile Beach, south of Broome (19° S). The KGBF is managed by limiting entry, gear restrictions, and seasonal and spatial area closures. Currently only seven licences access the KGBF. The total landings of barramundi from all four prescribed fishing areas within the KGBF were 59.6 t and 57.1 t for 2009 and 2010 respectively and are the highest recorded catches since 1987 which is primarily due to a large increase in effort during these years (Department of Fisheries 2011). Recreational catch of barramundi in the KGBF was last assessed in 2000 and represents 1-2% of the commercial catch (Department of Fisheries 2011).

Northern Territory

The commercial sector of the barramundi fishery in the Northern Territory operates from the high water mark to three nautical miles seaward from the low water mark and is restricted to waters seaward from the coast and river mouths. This fishery uses gillnets and has tight management controls that restrict the number of licences, areas and seasons fished as well as gear type and amount. Catches have varied in the commercial barramundi fishery over the last 37 years but effort has declined substantially which has resulted in some of the highest CPUE recorded in recent years (Figure 9.1). The major commercial fishing areas are the Van Diemen Gulf, East Arnhem Land, Anson Bay, Central Arnhem Land and Limmen Bight (Northern Territory Government 2011).
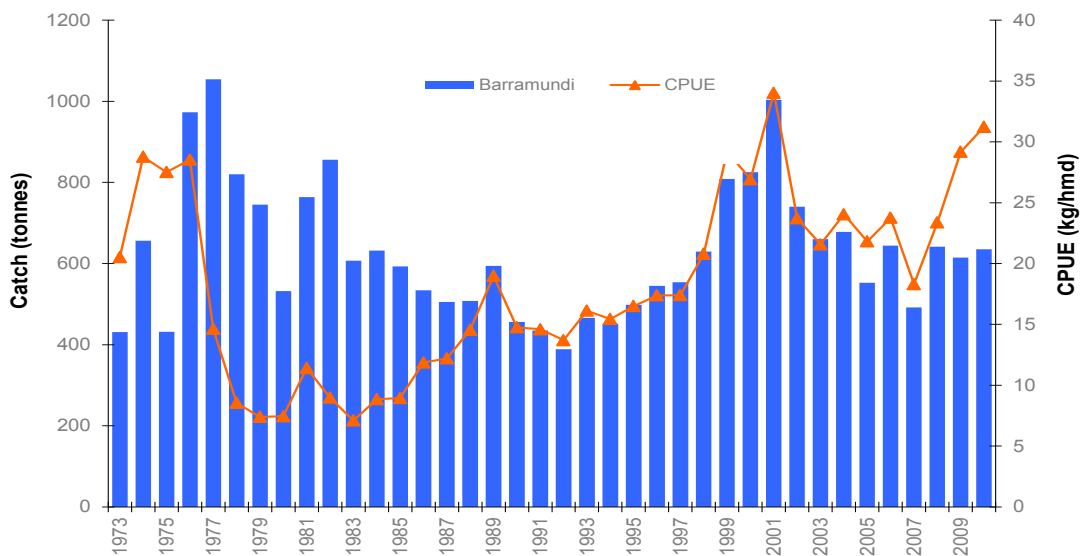
Figure 9.1. Catch and catch-per-unit-effort (CPUE) for the Northern Territory commercial barramundi fishery from 1973 to 2010.

Recreational anglers and Fishing Tour Operators (FTOs) also target barramundi using rod and reel in the same areas as the commercial sector and have gear restrictions, possession limits and seasonal area closures. All sectors in the Northern Territory have a minimum legal size of 55 cm total length (Table 9.1). An estimated annual harvest of 105,131 barramundi (~368,000kg) was recorded for the NT recreational fishery and 70% of the Indigenous barramundi catch in Australia is from the Northern Territory (Henry and Lyle 2003). Barramundi is the most targeted fish by recreational anglers in the Northern Territory (Coleman 1998). FTOs catch approximately 40,000 barramundi annually; FTOs and the recreational sector release between 70 and 90% of the barramundi they catch, with a high (~91%) post-release survival rate (de Lestang *et al.* 2004).

Table 9.1. Fisheries regulations for barramundi in respective jurisdictions of northern Australia.

Jurisdiction	Minimum legal size (total length in cm)	Maximum legal size (total length in cm)	Closure rules
Western Australia	55	Nil	na
Northern Territory	55	Nil	na
Queensland - Gulf of Carpentaria	60	120	October to January, variable on spawning moon
Queensland - East Coast	58	120	1st November to 1st February

While there were some concerns about overfishing in some of the more popular river systems in the 1970's and 1980's, current assessments of the barramundi stocks across the NT indicate that they are being harvested well within sustainability limits (Northern Territory Government 2011).

Queensland

In Queensland barramundi are taken as part of two commercial finfish fisheries: Gulf of Carpentaria Inshore and East Coast Inshore. In each fishery, specific fishing endorsements (i.e., licences) are required to harvest barramundi. These two fisheries are managed separately by limited entry, minimum and maximum size limits (Table 9.1), spatial closures (some of which allow recreational only fishing), temporal closures to protect spawning stock, and a recreational bag limit. Commercial catches of barramundi vary spatially and temporally (Figure 9.2) and can be significantly related to river flow or rainfall (Robins *et al.* 2005; Balston 2009a) and evaporation (Balston 2009a). Variability in catch probably represents changes in underlying stock abundance linked to environmental drivers, although in many studies there is still a significant amount of variation in catch that is unexplained (DEEDI 2010a&b).

Barramundi is a key species for recreational fishers. The recreational harvest of barramundi was estimated to be ~230 tonnes in 2000 (Henry and Lyle 2003). In addition, in 2005 the estimated harvest by recreational fishers in Queensland was 51 t (McInnes 2008). Barramundi, are a less significant part of the indigenous finfish harvest, compared to NT and was estimated to be ~5,745 barramundi in 2000 (Henry and Lyle 2003).
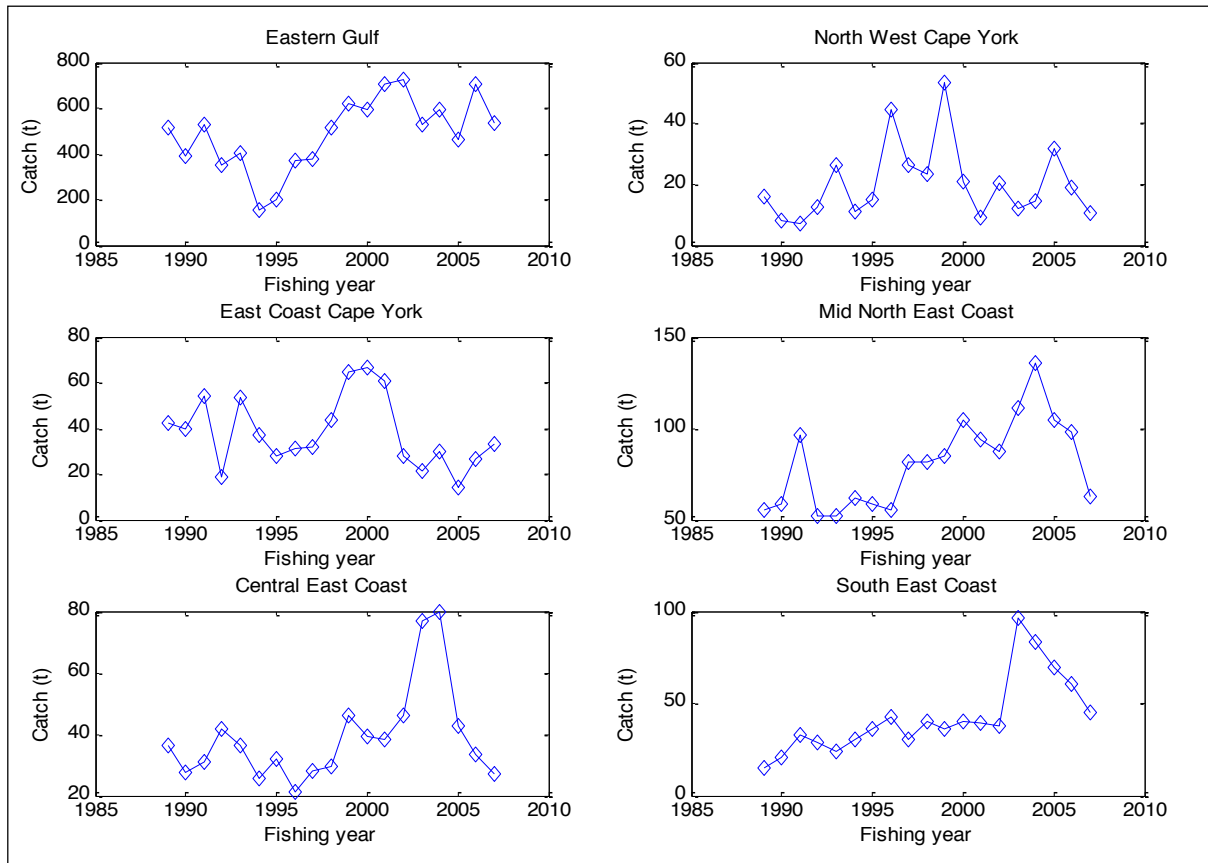
Figure 9.2. Regional commercial catch (tonnes) of barramundi in Queensland 1989 to 2007 (taken from Campbell *et al.* 2007). Eastern Gulf stock and North-west Cape York stock located in the Gulf of Carpentaria. All other stocks are located on the Queensland east coast.

Life History

> **Key points:**
> - Sub-stocks exist across northern Australia, usually associated with river systems, with limited exchange between sub-stocks.
> - Recruitment is highly variable and is correlated with seasonal rainfall or riverflow.

Life cycle, age, and growth

Barramundi have a complex and spatially variable life history, displaying non-obligatory catadromy i.e., migrating from freshwater to saltwater to spawn. The proportion of the population that migrates to freshwater habitats varies between catchments and within years within catchments (Pender and Griffin 1996; Milton *et al.* 2008; Halliday *et al.* 2012). The proportion of the barramundi population that accesses freshwater habitats probably depends on the variable accessibility of these habitats associated with seasonal rainfall. For example, in the perennially flowing Daly River, ~86% of estuarine adult barramundi had accessed freshwater habitats for a period of greater than three months (Halliday *et al.* 2012) compared to the Fitzroy River (Queensland east coast) where ~50% of estuarine adult barramundi had accessed freshwater habitats (Milton *et al.* 2008).

Barramundi are also protandrous hermaphrodites, initially maturing as males at two to five years then changing sex to females at between five and seven years, although a small proportion of the population are primary females and are mature at much smaller sizes and age (Moore, 1979; Davis 1984). Mature female barramundi are thought to reside in the lower reaches of estuaries and along the coastal foreshore (i.e., in saltwater habitats, Dunstan 1959). The life cycle of barramundi generally results in the spatial separation of male and female fish, with smaller and younger male fish residing in the upper estuary or in freshwater reaches of the river. Mature males must move downstream to the estuary in order to participate in spawning. Mature barramundi are thought to be stimulated to move downstream to areas of higher salinity by the first freshwater flow in spring that lowers the salinity of estuarine waters (Rod Garrett, pers. comm. 2000). This could be achieved by small freshwater flows that do not necessarily release landlocked individuals. Most barramundi participate in one or more spawning seasons as males before undergoing sexual inversion, becoming functional females by the next breeding season (Schipp *et al.* 2007).

In Australia, barramundi spawn during spring and summer. The timing and duration varies between regions, rivers and years, depending on water temperatures and lunar and tidal cycles (Table 9.2). Breeding takes place in high-salinity reaches of estuaries and nearby coastal foreshores. In general, spawning activity peaks during new and full moon periods (Grey 1987), as large incoming tides may help eggs to move into estuaries. Movement of adults to spawning areas is triggered by the seasonal increases in water temperature (Grey 1987). High salinity appears to be the main requirement of spawning grounds i.e., 32 to 38 ppt (Davis 1987; Rod Garrett, pers. comm. 2000). Gametogenesis in barramundi is initiated by seasonal increases in water temperate and photoperiod (Russell 1990). Each female commonly releases three to six million eggs which are pelagic, average 0.7 mm in diameter (Russell and Garrett 1985), and once fertilised will hatch in less than 24 hours at water temperatures of ~28°C (Schipp 1996; Griffin; Rod Garrett, pers. comm. 2000). Optimal hatching occurs at salinities between 20 and 30 ppt with lowered hatching success at salinities higher or lower i.e., 35 or 5-15 ppt (Maneewong 1987). Of the limited work that has been published on the optimal pH for hatching success, De (1971, cited by Pusey *et al*. 2004) reported that larval barramundi had a narrow pH range of 7.4 to 7.6 units. Barramundi larvae spend about three weeks in inshore waters and require high salinity water (Schipp 1996). The completion of the major part of the breeding cycle before the onset of the wet season is probably a strategy for eggs and larvae to avoid low-salinity water (Russell and Garrett 1985) and so that juveniles can take advantage of the aquatic habitat that results from rains in the monsoon season (Davis 1985).

Table 9.2. Spawning seasons of barramundi across northern Australia.

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

Post-larval barramundi move to available estuarine wetlands and flood plains as nursery habitats (Russell and Garrett 1985). Moore (1980) suggested that barramundi larvae are cued or attracted upstream by chemicals released from swamps. Peak spring tides and seasonal flooding assist barramundi post-larvae to enter supra-littoral habitats (Russell and Garrett 1985), coastal lagoons (Grey 1987) and other seasonal habitats that form during the monsoon season (Williams 2002). Coastal swamps (i.e., adjacent to the coast and estuary) form the predominant nursery habitat for post-larval barramundi in areas of northeastern Queensland where large river systems are absent (Russell and Garrett 1985). Monsoon rains also create a variety of temporary nursery habitats for juvenile barramundi that are highly productive in food resources and are thought to offer protection from larger predatory fish. These swamps rely on "flood rains" to connect with more permanent waters (Russell and Garrett 1985). Juvenile barramundi were reported moving into supra-littoral pools in the Fitzroy River estuary during March (Hyland 2002). Griffin (1985) suggests that rainfall replenishes the water levels in supra-littoral habitats between high tides (thereby maintaining these nursery habitats for longer periods) and that "the amount of time that the young of the year fish are able to utilise this safe and rich environment is limited by the amount and extent of rainfall during the wet season". Griffin (1985) only considers rainfall, although it is possible that floods that inundate flood plains may have a similar effect in extending the spatial and temporal extent of these high quality nursery habitats. This relationship was further confirmed with an additional two years of data, when Griffin (1987) reported a significant correlation (r^2=0.81) between juvenile abundance (i.e., young of the year) and early wet season rainfall. So not only is rainfall important for juvenile survival but also the timing of the rainfall.

Juvenile barramundi depart these habitats at the end of the wet season. Lowering of water levels and depletion of food in seasonal habitats is likely to stimulate juvenile barramundi to move to other habitats (Russell and Garrett 1985). For example, juvenile barramundi began moving from swamps in Trinity Inlet (Cairns) in April and remained in tidal creeks until December and January (Russell and Garrett 1985). In the Gulf of Carpentaria, floodwaters recede around March. Some juvenile barramundi move to permanent freshwater habitats when the seasonal coastal habitats dry-out (Russell and Garrett 1985); these individuals are moving upstream to freshwater habitats at about three to five months of age. In comparison, juvenile barramundi in Papua New Guinea waters take more than one year to reach inland freshwater habitats because of the need to migrate along the coast from spawning areas.

Where access to permanent freshwater permits, a varying proportion of the juvenile barramundi population migrates upstream, predominantly in spring and summer towards the end of their first year of life i.e., >9 months (Stuart 1997; Stuart and Mallen Cooper 1999). Otherwise juveniles remain in estuarine habitats and either access upstream habitats in their second year or remain in the estuary for their entire life (Russell and Garrett 1988; Pender and Griffin 1996). Barramundi mature at three to four years of age and then return to the estuary when conditions permit, to spawn alongside estuarine residents (Grey 1987).

Barramundi are capable of rapid growth, typically reaching 35 cm total length in their first year of life, 50 cm in their second year and 60 cm by the end of their third year (Griffin and Kelly 2001). Their asymptotic length is >150 cm, with a weight of up to 40 kg. Growth is seasonally variable (Xaio

2000) and probably reflects seasonal water temperature and food availability, as feeding activity is greatly reduced at water temperatures less than 24°C (Pusey *et al*. 2004). Growth is also spatially and temporally variable (Davis 1987), and is probably a reflection of environmental conditions. Growth variability is significantly related to the freshwater flows experienced by individuals (Robins *et al.* 2006), although other factors (e.g. genetic variation) are also likely to be important.

Variable growth may also account for the observed variable size-at-maturity. Davis (1982) reported that size-at-maturity for males was 60 and 55 cm for fish in the Northern Territory and South-eastern Gulf of Carpentaria respectively, and for females was 90 and 85 cm respectively. Davis (1982) went on to speculate that these "size differences were due to a slower growth rate of barramundi in the Gulf of Carpentaria, both processes being related to age rather than size. Griffin (1988) also speculated that growth rates differ between the Daly and Liverpool River (Northern Territory) based on differences in the size-at-age structure in the two rivers. Barramundi are a relatively long-lived species i.e., >20 years, with specimens of 32 years recorded from central Queensland (Staunton-Smith et al. 2005; Halliday et al. 2011). Longevity is likely to vary between regions, depending on environmental conditions and fishing pressure.
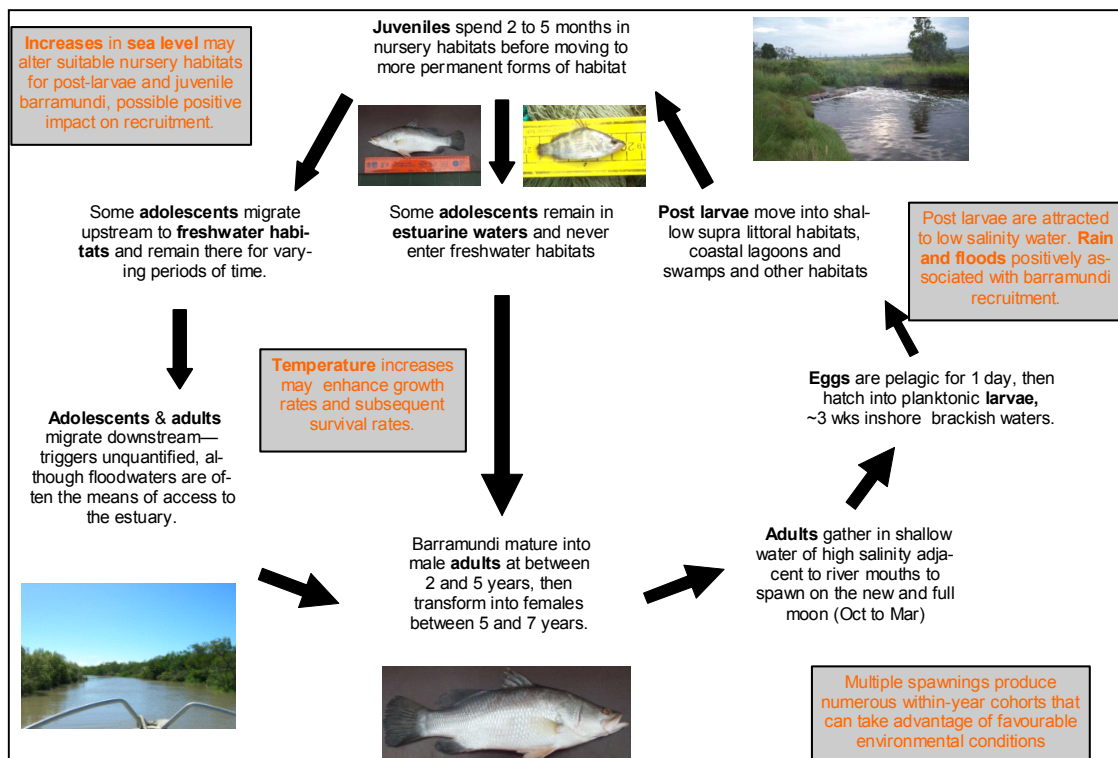
Figure 9.3. Summary of the life cycle of barramundi (*L. calcarifer)* and the points of exposure to relevant climate change drivers or known impacts. (Images sourced from QDAFF).

Distribution, habitat and environmental preferences

Barramundi are widely distributed throughout the tropical and sub-tropical waters of the Indo-Pacific region from the Arabian Gulf eastwards to China and Japan and southwards to Papua New Guinea and Australia (Pusey *et al.* 2004*).* In Australia, barramundi occur from the Ashburton River

(22°30' S) in the Kimberley and Pilbara regions of Western Australia northwards throughout the Northern Territory and Gulf of Carpentaria and down the Queensland east coast as far south as the Noosa River (26°30' S) (Figure 9.4; Schipp 1996).

Barramundi occur in a wide range of habitats including coastal foreshores, estuaries, tidal creeks, swamps, flood plains, coastal lagoons and upstream rivers where accessible from the sea. Juvenile and adult barramundi appear to be highly tolerant to a wide range of water acidity having been collected over a wide range of pH: 4.0 to 7.2 in the Alligator River region (NT); 6.1 to 9.12 in floodplain lagoons of the Normanby River (Qld); 5.2 to 5.6 in dune lakes of Cape Flattery region (Qld); and <4 in tidal creeks near Trinity Inlet, Cairns (Qld) (Pusey *et al.* 2004). Barramundi are more abundant in areas where there are large, slow flowing rivers and absent from areas without large river flows (Dunstan 1959). Barramundi occur in both clear and turbid waters. Temperatures appear to limit their distribution, with 15°C a critical lower thermal limit and 44°C a critical upper limit (Rajaguru 2002), although their optimum for growth and protein metabolism is 27 to 33°C (Katersky and Carter 2007).
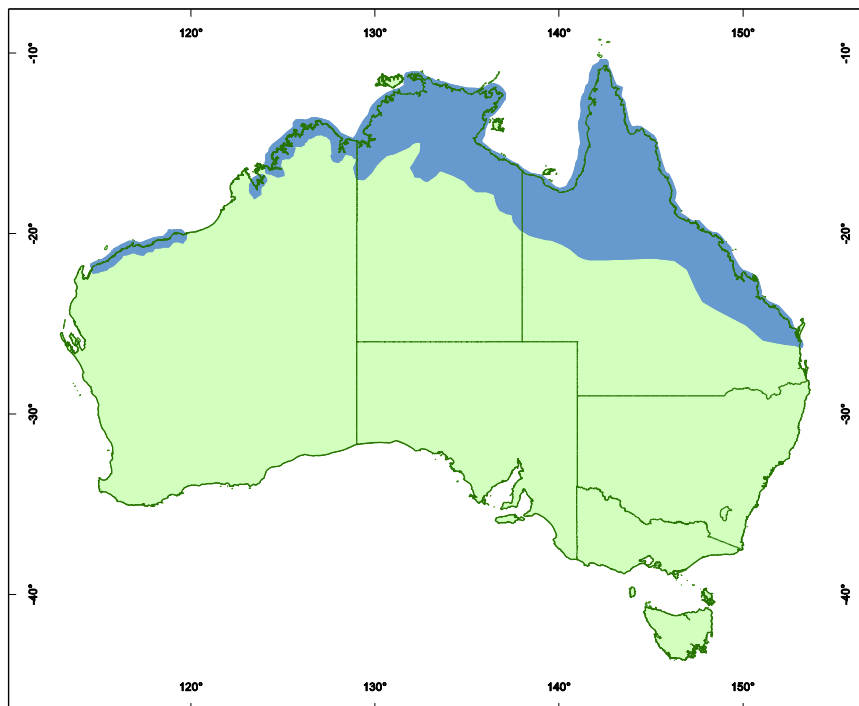
Figure 9.4. Distribution of barramundi in Australia.

Predators and prey

Barramundi have an important ecological role in tropical Australian estuaries (Dunstan 1959) as a large opportunistic ambush predator. Barramundi have an ontogenetic change in diet from insect larvae and micro-crustaceans to macro-crustaceans to fish (Davis 1987), which roughly corresponds to the size of organism that will fit into their mouth. In some circumstances, barramundi are also cannibalistic with young-of the year eaten by larger individuals of older cohorts (Russel and Garret 1985, Schipp 1996).

Recruitment

Barramundi recruitment is variable temporally and spatially (Griffin and Kelly 2001; Staunton-Smith *et al.* 2004; Halliday *et al.* 2011; Halliday *et al.* 2012). Successful recruitment of barramundi to the commercial fishery depends on the ability of post-larvae, juveniles and adults to migrate between the wetland, freshwater and marine environments suitable for each stage in the life cycle (Moore and Reynolds 1982). It also depends on the survival rate of each of these stages, which may be related to growth rates and freshwater flow (Robins *et al.* 2006). Griffin and Kelly (2001, p7) suggested "rainfall is an important influence (on recruitment), presumably through its effect on the availability and habitability of swamp habitat, particularly in the early part of the spawning season". Staunton-Smith *et al.* (2004) and Halliday *et al.* (2012) reported significant positive relationships between seasonal freshwater flows and the year-class strength of barramundi in five catchments in northern Australia (i.e., the Fitzroy, Mitchell, Flinders, Daly and Roper Rivers). Sawynok and Platten (2011) reported positive relationships between catch rates of recreationally caught 0+ barramundi in the Fitzroy River region (central Queensland) and rainfall and riverflow variables, with January rain having the highest r value (i.e., 0.56 p<0.01). They also reported a significant (step-wise backward) generalized linear model, that explained 50.6% of variation in 0+ catch rates that included wet season (Nov to Mar) flow (p=0.092), January flow (p=0.097), January rain (p=0.006) and February rain (p=0.103).

Barramundi stocks in northern Australia are genetically different between the Gulf of Carpentaria and the Queensland east coast (Shaklee and Salini 1985; Salini and Shaklee 1988; Williams 2002). Davis (1985, p189) suggests that because of localised spawning and genetic evidence of stock heterogeneity that "recruitment into major river systems would depend largely on the successful spawning of local populations" and that "the populations in different river systems may be quite independent of each other, and it may be appropriate to manage them as separate stocks". Tagging studies have demonstrated that while barramundi can move large distances between estuaries, most individuals remain within a specified region.

Current impacts of climate change

Key points:
- Barramundi populations are known to be reliant in many ways on rainfall and riverlow however there are no current impacts that can be attributed to climate change.

There are no documented current impacts of climate change on barramundi, although there are documented links between river flow/rainfall.

Sensitivity to change

Key points:
- Barramundi are sensitive to changes in rainfall and riverflow, which can influence catch, annual recruitment and growth rates.
- Predicting local impacts on populations is complex due to wide use of habitats during different life history phases, however; generally lower rainfall is likely to have negative consequences for populations.

There are several well documented strong relationships for barramundi between rainfall/riverflow and (i) catch (Robins *et al.* 2005; Meynecke *et al.* 2006; Balston 2009a, b; and Meynecke *et al.* 2011); (ii) recruitment (Staunton-Smith *et al.* 2004; Halliday *et al.* 2011, 2012; Sawynok and Platten 2011); and (iii) growth (Robins *et al.* 2006). Recent modelling of the possible effects of climate change on barramundi populations suggests that, on average, stock sizes and harvests would be reduced as a consequence of reduced river flows (Tanimoto *et al.* 2012).

A vulnerability assessment of Kakadu to climate change impacts found that barramundi were a key species that had "medium-high" risk of "decrease in abundance" by 2030 and 2070 (BMT WBM 2010). This was based on losses in nursery habitats as a consequence of sea level rise, although the report recognised possible increases in adult habitat, but reduced floodplain connectivity from reduced rainfall.

Sawynok and Platten (2011) suggested that the increase in the duration between large flood events may impact on strong recruitment years for barramundi. Currently, strong recruitment years are a feature of several regional stocks of barramundi (Halliday *et al*. 2012) and appear to drive the productivity of associated fisheries for several years. Sawynok and Platten (2011) then suggest that if the length of time between large recruitment events exceeds eight years, then there may be issues with the sex ratio of the spawning population with "uncertain consequences".

Resilience to change

> **Key points:**
> * Barramundi are likely to be resilient to increases in temperature projected for northern Australia over at least the medium-term (~50 years) as they have a wide thermal tolerance and are capable of large spatial movements.
> * Populations of barramundi are likely to be impacted by reduced riverflows, particularly by periods of extended drought.

Barramundi are likely to be resilient to climate change as they are adapted to a wide variety of habitats and temperature and salinity levels (Grey 1987) and are capable of large within catchment movements. However, populations associated with specific river catchments may suffer reductions in abundance as a consequence of potential reductions in important freshwater habitat (squeezed by sea level rise).

Other

> **Key points:**
> * Water resource extraction/management (particularly on the Queensland east coast) is a potential additional stressor of the estuarine ecosystem, particularly through reducing the connectivity of floodplains to downstream ecosystems.

Ecosystem level interactions

Barramundi are a key predator in tropical river systems. They take advantage of seasonally available food resources in both estuarine and floodplain habitats (Salini *et al*. 1990; Jardine *et al*. 2011). Predation by barramundi is an important factor that determines the structure of the fish assemblage

in many upstream habitats (B. Pusey unpublished data). Factors that influence the productivity of the lower food web will impact on barramundi populations.

Additional (multiple stressors)
Barramundi production is linked to river flows and the connectivity of floodplains to estuaries (Jardine *et al. 2011*). Management of water resources for human use has the potential to exacerbate climate stressors, particularly under scenarios with reduced rainfall as human demand for water resources often takes precedent over ecosystem needs.

Critical data gaps and level of uncertainty
It is relatively well documented that variability in abundance in barramundi populations in northern Australia is linked to variation in rainfall and river flow (Halliday *et al.* 2011, Halliday *et al*. 2012) and is dependent on floodplain connectivity and productivity (Jardine *et al.* 2011). What is not well understood is how these systems will respond to the changing climate.

Acknowledgements

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10.   Barred Javelin, *Pomadasys kaakan*

Authors: Richard J. Saunders, Natasha Szczecinski and David J. Welch

A juvenile barred javelin, *Pomadasys kaakan.* (Image sourced from JCU).

The Barred Javelin, *Pomadasys kaakan*, is a member of the family Haemulidae (the grunters). The species occurs throughout the Indo-Pacific from the Red Sea and the east coast of Africa to south-east Asia and northern Australia (Froesy & Pauly 2012).

The fisheries

- Commercial catches of barred javelin are generally reported as "grunter". This is a complex containing at least three species.
- The barred javelin is an important by-product species in commercial fisheries targeting barramundi
- The species is an important recreational fishery species, particularly in north Queensland

Western Australia
25 t of grunter (*Pomadasys* spp.) were landed across all of WA's commercial fisheries in 2009/10 financial year (Department of Fisheries 2011). Although this complex includes *P. kaakan*, *P. argenteus* and *P. maculatus* the latter is likely to be a minor component given their small size. *P. kaakan* has been identified as one of the top 20 species landed by recreational fishers in the Pilbara and West Kimberley but the size of the catch in the region has not been estimated. Some fine scale regional data is available on catch in north-west WA (Newman et al. 2009).

Northern Territory
Grunters are taken as by-product in commercial barramundi fisheries in the NT. However, less than 2 t of grunter have been reported each year for 2008, 2009 and 2010 (NT Government 2011). No data on the importance of *P. kaakan* in recreational fisheries in the NT is available however most of the NT recreational catch comes from the Gulf of Carpentaria (Thor Saunders, pers. comm.).

Queensland

In Qld, commercial fishers land grunter in the Gulf of Carpentaria Inshore Fin Fish Fishery (GOCIFFF) and the East Coast Inshore Fin Fish Fishery (ECIFFF). Average catch in the GOCIFFF over the past seven years was 27 t (to 2009) (DEEDI 2011a). In the ECIFFF average catch was 28 t over the past four financial years (to 2009/10) (DEEDI 2011b). The annual catch by charter operators in the ECIFFF has ranged from 401 kg to 2,288 kg since 2004. The status of the species in Queensland is listed as uncertain due to poor knowledge of the recreational harvest.

The species has been noted as one that is a common target species by recreational fishers (Greiner & Patterson 2007; Hart & Perna, 2008). The tourist recreational catch of *P. kaakan* in the Gulf of Carpentaria was estimated to be between 100 and 118 tonnes over the period March – September 2006. Further, at a local scale, the Karumba recreational tourist fishery (from May to August inclusive) catch of *P. kaakan* was 13.5 t, representing 30% of the total catch in that fishery (Hart & Perna, 2008).

Recent research work in the Lucinda region of the north Queensland east coast indicates that recreational fishers catch fish predominantly between 280 and 360 mm TL whereas commercial fishermen catch a more even spread of sizes with significantly more fish over 600 mm than recreational fishers (Szczecinski, unpublished data). Furthermore, the recreational catch of grunter was highly skewed toward females with a ratio of 15:1. In the commercial sector however, this ratio was only 2:1 (Szczecinski, unpublished data).

Life history

- Barred javelin occupy estuarine and nearshore habitats across tropical and sub-tropical Australia.
- They mature at a small size by 3 years of age and in many parts appear to have a protracted spawning season lasting much of the year.

Life cycle, age and growth

The life history of *P. kaakan* is poorly understood as very limited research has been done. However, on the Queensland east coast the reproductive period off Townsville is reported to occur between September and November (Bade 1989). Further south on the Qld east coast, a more extensive reproductive period, from September to March, has been reported (Russell 1988). Recent research in the Lucinda region in far north Queensland also indicates an extremely protracted spawning season with actively spawning fish collected from August to June, although fish were not collected during January, April, May and December (Szczecinski, unpublished data). The species is thought to mature by its third year (Garrett, 1996). In the Lucinda region, over 50% of fish (males and females) are mature by 200-239 mm TL (Szczecinski, unpublished data). Histological sections of mature ovaries indicate the species is most likely a multiple batch spawner (Bade, 1989). Frosey & Pauly (2012) note that spawners form shoals near river mouths during the winter, but the statement is unreferenced.

Age and growth of *P. kaakan* were described for the Queensland east coast by Garrett (1996). Von Bertalanffy growth parameters reported were L_∞ = 579 mm FL, K= 0.35 and t_0= -0.66. More recent information on growth has been determined for the species in the Lucinda area (near Hinchinbrook Island, Far North Queensland) and this data differed from that of Garrett (1996) with Von Bertalanffy parameters L_∞ = 746 mm FL, K= 0.18 and t_0= -0.79 (Szczecinski, unpublished data). The oldest fish reported was 14 years (Garrett 1996), estimated from increments in whole otoliths. *P. kaakan* is reported to reach 800 mm (Froesy & Pauly 2012) but in Bade (1989) the largest fish reported was 530mm TL, and 610mm FL in Garrett (1996).
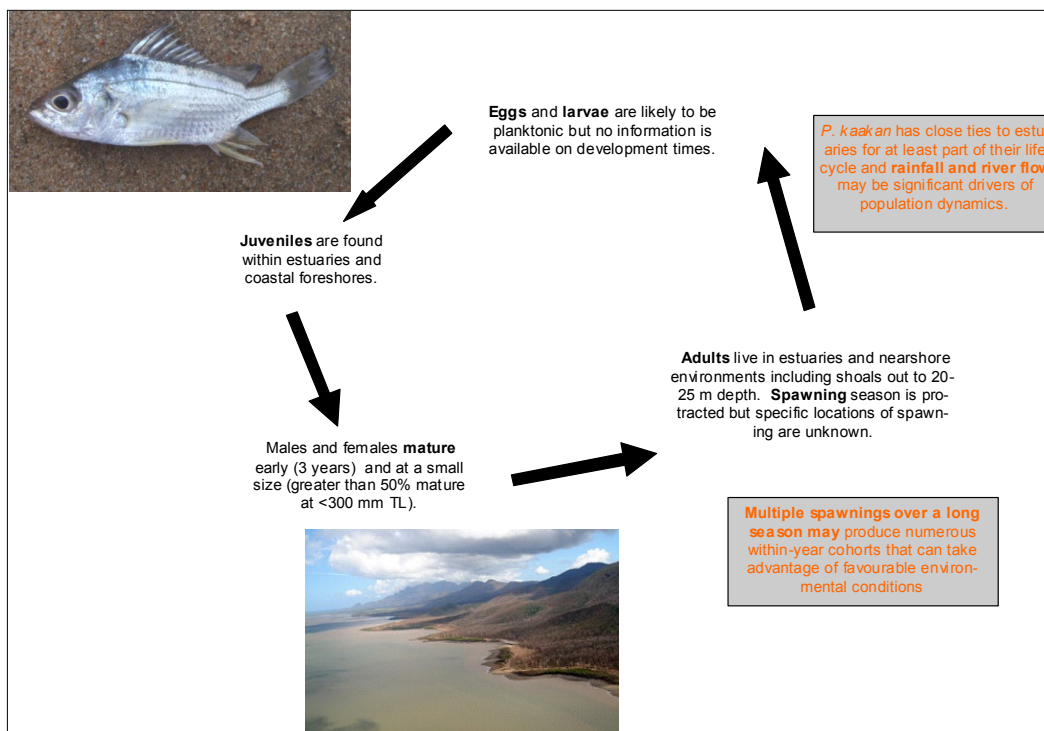
Figure 10.1. Generalised life cycle of the barred javelin, *P. kaakan*, and the stages of potential environmental driver impacts.

Distribution, habitat and environmental preferences

P. kaakan occurs throughout the Indo-Pacific from the Red Sea, east coast of Africa, south-east Asia and northern Australia (see Figure 2) (Froesy & Pauly 2012). It lives inshore primarily in estuarine and shallow coastal waters (Bade 1989; Smith & Heemstra 1986).

Predators and prey

The diet of *P. kaakan* around Townsville on Australia's east coast was described in Bade (1989). Principal prey items identified from stomach contents were polychaetes, crustaceans and fishes (Bade 1989). In that study, the most common prey item for larger fish (over 150mm) were decapods, while polychaetes were the most common prey item for specimens under 150mm.

Recruitment

There are no known measures of recruitment of grunter species in Australia and no population age structure information is available.
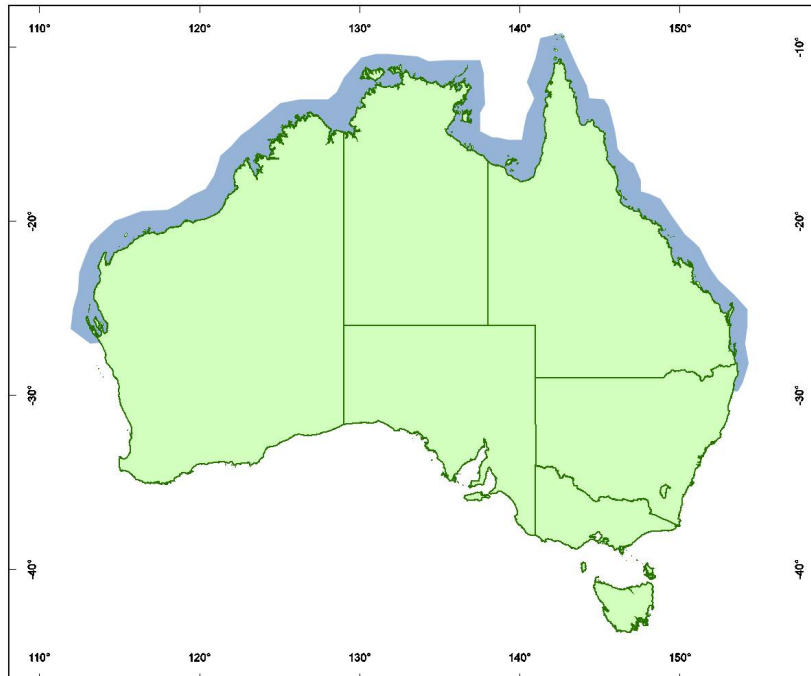
Figure 10.2. The Australian distribution of barred javelin.

Current impacts of climate change

There are no known current impacts of climate change on grunter species in Australia.

Sensitivity to change

- Sensitivity of barred javelin to environmental change is unknown.

The sensitivity of barred javelin to changes in environmental conditions is not known. However, they occupy nearshore and estuarine habitats and environments that are subject to large fluctuations in variables such as salinity, temperature and nutrient levels. As such, they are likely to be resilient to changes. It is also possible that rainfall and river flows are significant drivers of population recruitment and growth rates given this has been found to occur in several other nearshore/estuarine species (Halliday et al., 2008; Meynecke et al., 2006; Robins et al., 2006; Staunton-Smith et al., 2004).

Resilience to change

- Likely to be resilient due to their widespread distribution covering varying nearshore dynamic habitats.

Barred javelin are distributed widely across northern Australia occupying many different tropical regions in environments known to vary widely (see above). They are therefore likely to be resilient to changes in the environment. They also have habitat to the south of their current range that they could occupy with increasing marine temperatures.

Other

- Due to little research historically there is a need to understand better the sensitivity of barred grunter to changes in climate-related variables.
- There is also a significant recreational fishery across parts of northern Australia however catcah estimates are lacking making it not possible to make statements about fishery sustainability.

Ecosystem level interactions

As with many other similar species the ecosystem level interactions of barred javelin under climate change are very difficult to predict given uncertainty in exposure and sensitivity as well as predation and competition.

Additional (multiple) stressors

Fishing impacts are probably low at current levels in most regions of tropical Australia, however there remains a high level of uncertainty in the level of recreational harvest and the sustainability of this catch, particularly in the GoC and where recreational catches may be excessive (Hart and Perna, 2008). The stock structure of *P. kaakan* is unknown and will determine their sensitivity to localised depletions under fishing pressure or other impacts. Finally, barred grunter occupy estuaries and nearshore environments throughout their life cycle and are therefore exposed to land-based impacts such as water quality, agricultural and mining run-off, etc.

Critical data gaps and level of uncertainty

Recreational harvest levels is a key concern (Greiner & Patterson, 2007; Hart & Perna, 2008) and uncertainty for barred grunter in northern Australia and better estimates are needed for future more robust assessments of populations. Better understanding of the sensitivity of the barred grunter (and the spotted grunter, *P. argenteus*) to environmental variables such as temperature, salinity, pH and rainfall/river flow is needed to make more robust predictions about the potential impacts of climate change on these species.

Acknowledgements

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11.   Black Jewfish, *Protonibea diacanthus*

Authors: Thor Saunders and Emily Lawson

A recreationally caught black jewfish. (Image sourced from Jenny Ovenden).

The Fishery

Key points:
- The Northern Territory Line fishery is the only northern Australian fishery that takes significant quantities of black jewfish.
- They are likely to be overfished on the Queensland east coast.

Western Australia

The Kimberley Gillnet and Barramundi Managed Fishery (KGBF) currently take a small catch of black jewfish as a byproduct species. The total catch of black jewfish from the KGBF in 2010 was 4.3t (Department of Fisheries 2011). The recreational catch of black jewfish was estimated at 2-10% of the commercial catch in 2000. The Indigenous catch of black jewfish is unknown but is unlikely to be high.

Northern Territory

The Coastal Line Fishery of the NT is the only fishery in northern Australia that takes significant numbers of this species. This fishery operates in the near-shore waters and harvests a wide range of species, predominantly using hook and line gear. The fishery comprises commercial, recreational, Fishing Tour Operator (FTO) and Indigenous sectors and mainly targets black jewfish (*Protonibea diacanthus*) and golden snapper (*Lutjanus johnii*) (Phelan *et al.* 2008a, Northern Territory Government 2011). Black jewfish annual catch in this fishery has almost always been over 100 t (Figure 11.1). At the point of first sale in 2010, the catch value of the commercial sector of the fishery was $0.43 million of which black jewfish comprised $0.37 million (Northern Territory Government 2011).

Recreational and FTOs also target black jewfish although these sectors tend to catch substantially more of this species in the NT compared to Qld and WA. Recreational fishing surveys indicate that black jewfish catches by this sector are at least equivalent to the commercial harvest and substantially more when FTO catches are included. There is no size limit for black jewfish in the NT however various personal possession limits are in place to help regulate the impact of the recreational fishing sector. Presently, the recreational possession limit for black jewfish is two.
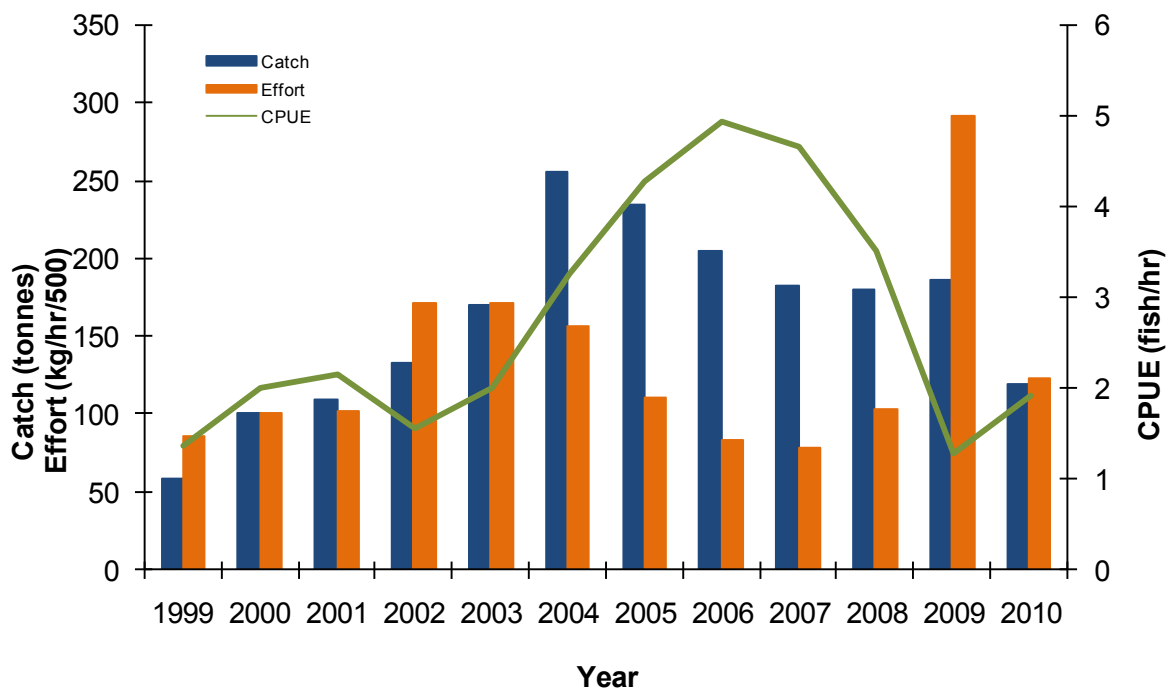
Figure 11.1. Catch and effort data of black jewfish from the commercial Coastal Line Fishery of the NT 1999-2010.

Queensland

The N3 inshore net fishery of the Gulf of Carpentaria Inshore Fin Fish Fishery (GOCIFFF) takes a small amount of black jewfish as a byproduct species with 9t caught in 2009. In addition, charter operators take a small quantity of black jewfish in the GOCIFFF with 157 kg caught in 2008 of which 67 kg was

released (DEEDI 2010). On the east coast the commercial catch is insignificant and the recreational harvest is unknown, however anecdotal reports suggest that this species has been overfished. Indigenous harvest has also shown to be significant in waters off Cape York (Phelan 2002).

Life History

> **Key points:**
> - Black jewfish grow and mature quickly.
> - Black jewfish are highly aggregative and suffer significant mortality from barotrauma-related injuries when caught in deeper waters.
> - Juveniles inhabit coastal bays and estuaries suggesting that recruitment may be influenced by coastal climatic factors.

Life cycle, age and growth

The black jewfish is a member of the Sciaenid family, which are also known worldwide as croakers or drums due to the distinct drumming noise they make using their swim bladder. Black jewfish grow fast, reaching almost 60 cm in their first year and 90 cm in their second, and live up to 13 years (Phelan and Green 2008). Phelan and Errity (2008) found that black jewfish in NT waters grow faster and spawn at different times than conspecifics in northern Queensland, despite similarities in both latitude and environmental conditions at the aggregation sites. Fifty per cent of black jewfish are sexually mature at 89 cm or around two years of age (Northern Territory Government 2011). In the coastal waters of the NT reproductive activity occurs during an extended season from August to January with peak spawning activity occurring in December (Phelan and Errity 2008). Black jewfish are known to form large aggregations during spawning making them vulnerable to capture during this time.

Black jewfish suffer significant barotrauma related injuries when captured at depth. From research surveys fish retrieved from less than 10 m were likely to survive if handled and released appropriately. However, 48% of fish caught at 10–15 m were likely to die when released and all fish landed from deeper than 15 m were likely to die when released (Phelan *et al.* 2008c).
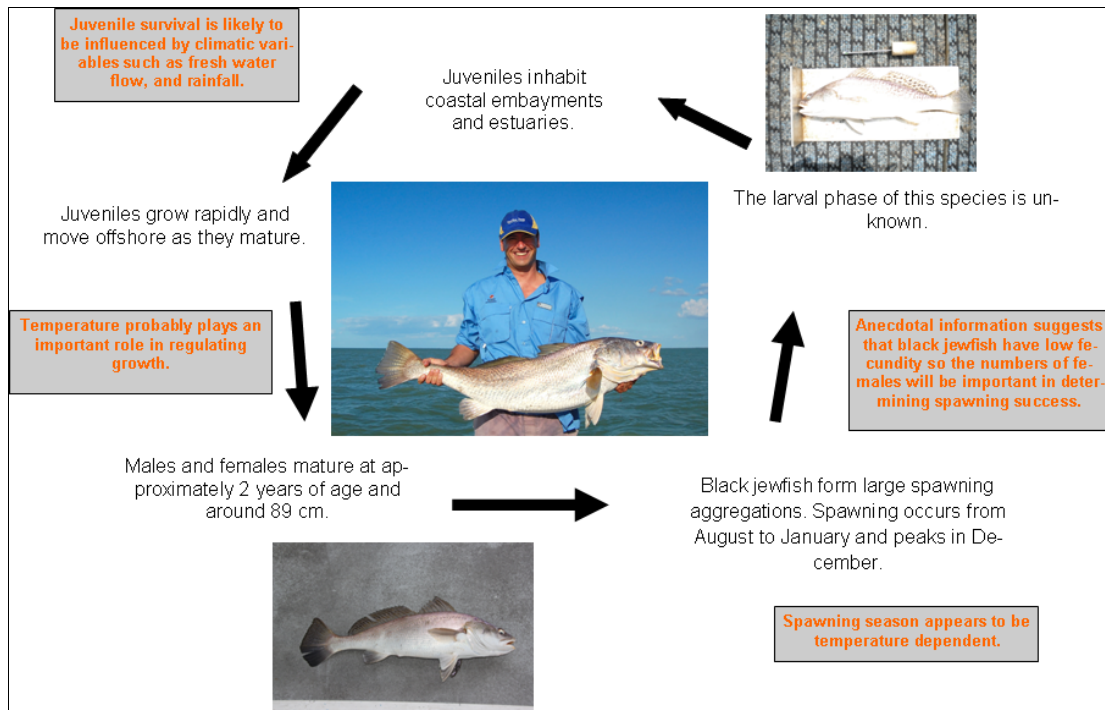
Figure 11.2. Summary of the life cycle of black jewfish and the points of exposure to relevant climate change drivers or known impacts.

Distribution, habitat and environmental preferences

Black jewfish is a migratory species found in turbid coastal waters throughout the Indo-West Pacific (India, Sri Lanka, Mayanmar, the Malay Peninsula, Thailand, Indonesia, Northern Australia, the Philippines, China and Japan). Adults tend to occupy near shore reefs (although they do occur in deeper waters offshore) while juveniles tend to inhabit coastal embayments and estuaries (Hay *et al*. 2005). In northern Australia waters black jewfish occur from central eastern Queensland to northern Western Australia (Figure 11.3; Newman 1995, Phelan 2008).
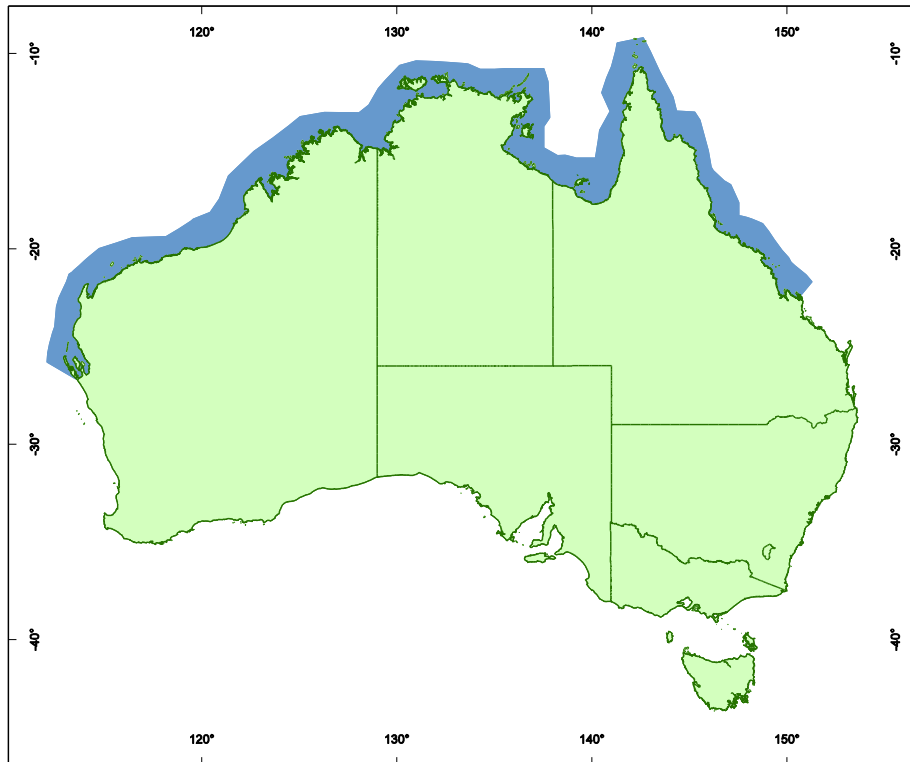
Figure 11.3. Australian distribution of black jewfish.

Predators and prey

Adult black jewfish are an opportunistic carnivore that preys on crustaceans, octopus, squid and fish (Hay *et al.* 2005) whereas juveniles are likely to feed on smaller crustaceans and fish due to their smaller size and different habitat. Juvenile black jewfish are likely to be preyed upon by large coastal fish such as barramundi or larger conspecifics and sharks.

Recruitment

Spawning takes place between August and January and peaks in December and January in the NT (Phelan and Errity 2008) and between April and September in Cape York (Phelan 2002). The factors that influence recruitment success are poorly understood although it is likely that abundance of spawning females, and coastal environmental drivers such as rainfall and river flow are important.

Current impacts of climate change

There are no known current impacts of climate change on black jewfish.

Sensitivity to change

Key points:
- The sensitivity of black jewfish to changes in environmental variables is poorly understood however it is highly likely that rainfall and riverflow are important given their life cycle.

The impact of climatic variables on this species is poorly understood. Given that juveniles mainly inhabit coastal estuaries and embayments, rainfall is likely to influence food availability and as a result growth and survival.

Resilience to change

> **Key points:**
> - High mobility and an extended spawning season provides some resilience to this species however they appear to be prone to overfishing as this appears to have occurred on the Queensland east coast.

Adults of this species are likely to be resilient to changes in climatic variables since they predominantly inhabit the marine environment and are capable of moving significant distances and occupying a range of depths/habitats. The protracted spawning period of this species also provides some resilience to environmental changes that produce unfavourable spawning conditions.

Other

> **Key points:**
> - Barotrauma related mortality is likely to cause additional pressures on populations, particularly those close to population centres where high levels of recreational fishing occur.
> - The linkage between black jewfish abundance and environmental factors, particulary rainfall/riverflow, is poorly understood.

Ecosystem level interactions
While this species is a large higher order predator in the tropics it is unlikely changes in abundance will significantly impact ecosystem function. Seasonal changes in productivity, and the factors that influence this, may be significant drivers of annual recruitment success.

Additional (multiple) stressors
The predictable aggregating behaviour of this species makes them vulnerable to targeted fishing (Semmens *et al.* 2010). Even sectors practicing catch and release whilst targeting these aggregations, are probably killing most fish they catch because of their sensitivity to barotrauma related injuries. A high level of fishing is therefore capable of rapidly removing a significant proportion of spawning adults and reducing egg production (Sadovy and Domeier 2005). Selective fishing of these aggregations may also truncate the size and age structure through targeting of larger fish (Sala *et al.* 2001), leaving the population less fecund (Eklund *et al.* 2000, Sala *et al.* 2001), and may alter genetic composition (Smith et al. 1991) and skew the sex ratio (Phelan *et al.* 2008c, Semmens *et al.* 2010).

Critical data gaps and level of uncertainty
There is very little known about the linkages between variation in environmental factors and black jewfish abundance. While the aggregative nature of this species is well documented their stock structure across northern Australia is unknown. In addition, the general biology and ecology of this species is poorly understood.

Acknowledgements

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12.   Black tip sharks, *Carcharhinus tilstoni* & *C. limbatus*

Authors: David J. Welch, Alastair V. Harry, Thor Saunders and Emily Lawson

A blacktip shark off the NSW coast. Photo: Pascal Geraghty.

Black tip sharks in Australian waters are comprised of two co-occurring species that are morphologically indistinct, making identification virtually impossible in the field. The two species are the common blacktip shark, *Carcharhinus limbatus*, and the Australian blacktip shark, *C. tilstoni*. Recent research has developed a genetic assay test to distinguish between the two species, however this is complicated by recent evidence of widespread hybridisation occurring between the two species in northern Australian waters (Morgan *et al.*, 2011; 2012). Vertebral counts and reproductive ecology has also been shown to be able to potentially distinguish the two species (Harry *et al.*, 2012), and more recently some key morphometric measurements have been demonstrated to distinguish between the two species with a 96% accuracy (Grant Johnson, unpublished data). Blacktip sharks have dominated commercial shark fisheries catches in northern Australia for the past 40 years (Stevens and Wiley, 1986; Harry *et al.*, 2011).

The fishery

> - Blacktip sharks are commercially important across most of northern Australia but tend not to be targeted by other sectors.
> - Highest catches are taken in the Northern Territory, the Gulf of Carpentaria and the east coast of Queensland.
> - Catches are quite variable and appear to be driven by market demand rather than environmental drivers.

Western Australia

The northern shark fisheries are the main fisheries targeting blacktip sharks in Western Australia (WA). These fisheries comprise the state-managed WA North Coast Shark Fishery (WANCSF) in the Pilbara and western Kimberley, and the Joint Authority Northern Shark Fishery (JANSF) in the eastern Kimberley. These fisheries historically used demersal longline with a small amount of pelagic gillnetting in the JANSF. Because of their similarities the northern shark fisheries are considered as a single fishery. Due to recent declines in sandbar shark (*C. plumbeus*) catch this fishery is tightly managed by; limited entry, substantial gear limitations and restricted access to a few areas within the fishery. Annual blacktip shark catch averaged 67 t in this fishery from 2006 to 2008 and, despite declines in other shark species, blacktip catches have remained stable over time, although their status is uncertain (Department of Fisheries, 2011). The recreational, charter and indigenous take of blacktip sharks in the northern shark fishery is unknown but is likely to be negligible because of the isolated nature of this coastline.

Northern Territory

The commercial Offshore Net and Line Fishery (ONLF) targets blacktip sharks along with grey mackerel. The fishery operates from the high water mark to the boundary of the Australian Fishing Zone (AFZ), although most of the effort occurs within 12 nautical miles (nm) of the coast. The fishery is managed by limited entry (17 licences permitted to operate), individual transferable effort allocations and strict gear specifications that facilitate the selective targeting of smaller, more productive sharks species, with a lesser impact on larger, less productive shark species. The fishery is managed by the Northern Territory (NT) Fisheries Joint Authority (NTFJA), in accordance with the *NT Fisheries Act 1988*.

Blacktip sharks were reported as a single group (*C. limbatus, C. sorrah* and *C. tilstoni*) until 1998. During this time blacktip shark catches increased from almost nothing to 670 tonnes in 1996 before declining to 266 tonnes in 1998. Thereafter, *C. limbatus* and *C. tilstoni* catches were reported separately and have increased from 104 tonnes in 1999 to 337 tonnes in 2010 and have remained stable above 300 tonnes for the last three years (Figure 12.1). In 2010 at the point of first sale the black-tip shark component of the fishery was valued at $0.83 million.
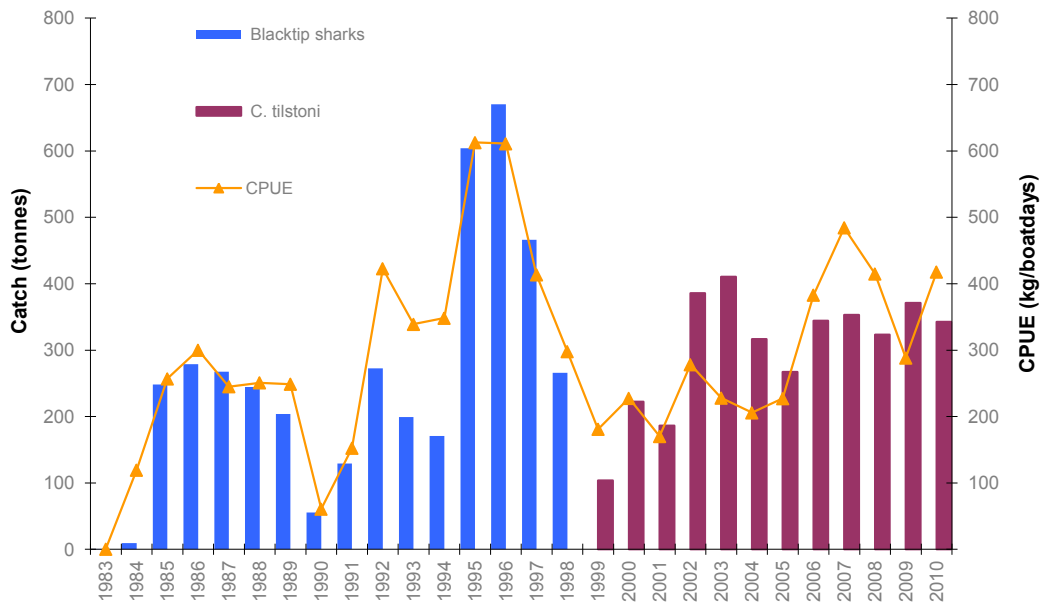
Figure 12.1. Northern Territory Offshore Net and Line Fishery catch and catch-per-unit-effort (CPUE) of blacktip shark for the years 1983-2010.

Sharks are generally not targeted by recreational fishers or Fishing Tour Operators (FTOs) in the NT, but are caught during other targeted fishing activities. In 2000-01, a survey of recreational fishers found that over 76,000 sharks were caught, with 8,000 harvested and the remainder released (Coleman, 2004). FTOs do not report sharks accurately by species. However, in 2010, FTOs caught 5,274 sharks and released 5,166 (98%). Currently individual recreational anglers are only permitted to retain two sharks (Northern Territory Government, 2010).

Queensland

Gulf of Carpentaria

The Gulf of Carpentaria Inshore Fin Fish Fishery (GOCIFFF) comprises inshore (N3) and offshore (N9) commercial net components, commercial bait netting (N11) and recreational, Indigenous and charter boat fishing within the Queensland jurisdiction of the Gulf of Carpentaria (DEEDI, 2012). The N9 net fishery harvests the most blacktip sharks (*C. limbatus, C. sorrah* and *C. tilstoni*) in the GOCIFFF and operates between 7 and 25 nm offshore. Smaller numbers of blacktip sharks are harvested in the N3 fishery, which mainly targets barramundi. Both net fisheries are authorised to use set mesh nets but are restricted by limited entry, allowable net length and drop and mesh size (DEEDI, 2012).

The recent historical annual commercial catch of all sharks in the Gulf of Carpentaria has been between approximately 300 and 650 t (Figure 12.2). However reporting by species in logbooks was only introduced in 2007 and so blacktip shark composition could not be determined until 2008 (DEEDI, 2012). For the years 2008 – 2010 blacktip sharks comprised an average of 68 % of the total catch.

Recreational fishers primarily use hook and line to catch target fish species and sometimes catch sharks as bycatch. In the most recent recreational survey (2005) only the total catch of shark was estimated so the catch of blacktip shark is unknown. Similarly, although charter operators have begun to report blacktip sharks separately, they generally only report total harvest of sharks and so blacktip composition is uncertain. Between 2004 and 2010 the charter sector reported catching on average 1,126 sharks with only one shark retained on average each year (DEEDI, 2012). There are no estimates of the Indigenous catch of shark in the Gulf of Carpentaria. The status of sharks in the Gulf of Carpentaria is 'undefined' due to lack of data.

East coast

The East Coast Inshore Fin Fish Fishery (ECIFFF) comprises multi gear commercial fisheries and recreational, charter and Indigenous fishing within all Queensland waters outside of the Gulf of Carpentaria (DEEDI, 2011). In the ECIFFF sharks are targeted by the commercial sector away from the coastline but generally within a few nautical miles. Nets account for 95% of the shark catch with line fishing taking the other 5%. In 2009, a Total Allowable Commercial Catch (TACC) of 600 tonnes, including rays, was introduced. In addition, tighter management arrangements were enforced, included limited entry into the fishery with licensees granted a fisheries 'S' symbol, and greater species resolution for shark species in logbook catch reporting (DEEDI, 2011). Although species reporting was done prior to 2009 the accuracy of identification is likely to improve after several years of reporting using the more detailed logbooks.
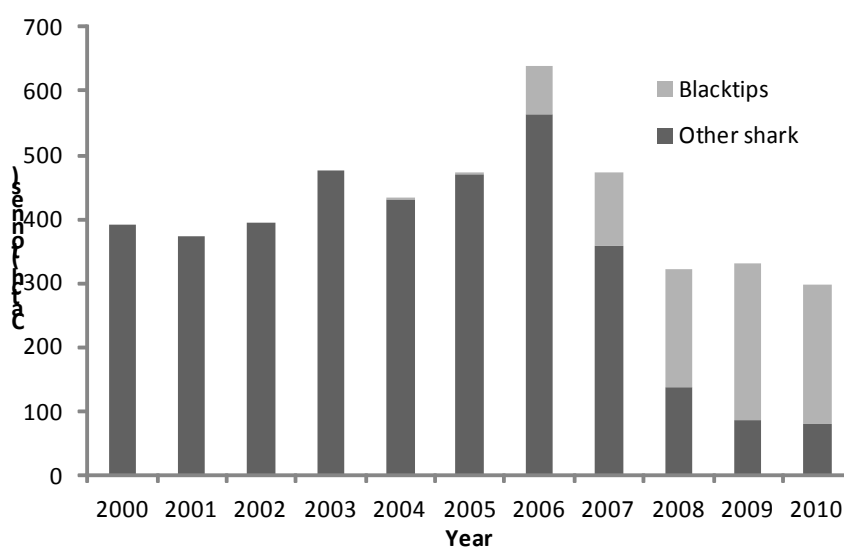
Figure 12.2. Reported commercial catch of sharks in the Gulf of Carpentaria inshore fishery (includes N3 and N9 sectors and net and line combined) for the years 2000 – 2010. The composition of blacktips in the catch is indicated and is not considered representative until at least 2008 due to compulsory reporting by species introduced in 2007. (Source: DEEDI, 2012).

The reported commercial catch of shark for the three years prior to the introduction of the TACC was 996 t, 1086 t, and 996 t respectively. During these years the blacktip shark (*C. limbatus, C. sorrah* and *C. tilstoni*) catch averaged 226 tonnes. The effect of management changes was a

reduction in total shark catch to 475 t in 2009-10 of which 36 % was reported to be blacktip shark (171 tonnes) (DEEDI, 2011). Blacktip sharks comprise a greater proportion of the total shark and ray catch in the northern region (41%) compared to the southern region (26%) of the fishery (DEEDI, 2011).

Sharks are not identified to species by recreational and charter operators so blacktip shark catch is unknown. The most recent estimates of recreational shark catch estimate that in 2002 there were 212 individual sharks harvested and 1,750 released, while in 2005 there were 104 harvested and 1,345 released (DEEDI, 2011). The reported catch (and release) of shark by the charter fishing sector in 2009-10 was less than 1 t. There are currently no estimates of Indigenous catch of shark in this fishery. The status of sharks in the ECIFFF is unknown and no assessment has been done due to lack of data (DEEDI, 2011).

Life history

- The common blacktip is found in tropical/sub-tropical waters globally while the Australian blacktip is endemic to tropical/sub-tropical Australia.
- The common and Australian blacktip species have different life history characteristics meaning effects of harvest are likely to be different for each species.
- The Australian blacktip is more productive and likely to be more resilient than the common blacktip shark.

Life cycle, age and growth
The common blacktip shark (*C. limbatus*) and the Australian blacktip shark (*C. tilstoni*) are are virtually indistinct species morphologically. Distinguishing between the two species is nearly impossible in the field and currently relies on using genetics, vertebral counts or reproductive ecology (Morgan *et al.*, 2011, 2012; Harry *et al.*, 2012). However, ecologically they are quite different species.

Australian blacktip shark
C. tilstoni give birth around January in northern Australia while on the east coast it appears to be slightly earlier in December, although this may vary from year to year (Stevens and McLoughlin, 1991, Stevens *et al.,* 2000; Harry *et al.*, 2012). In northern Australia, the usual size at maturity for *C. tilstoni* is 105 to 115 cm for males and 120cm for females, although females are not in maternal condition until 130 cm (Stevens and Wiley 1986). On the east coast of Australia maturity occurs at a slightly larger size; 120cm for males and 125cm for females although females are not in maternal condition until 138cm (Harry *et al.* 2013). Mating occurs in February-March with ovulation in March-April. The gestation period is 10 months and individuals breed each year. The average litter size is three to four and the size at birth is approximately 60-62 cm (Figure 12.3; Table 12.1) (Stevens and Wiley, 1986; Harry *et al.*, 2012; Harry *et al.* 2013).

Growth is relatively rapid in the first year of life: vertebral ageing indicated 17 cm growth in total length (TL) for *C. tilstoni* during the first year after birth. By the time the sharks are 5 years old, growth has declined to 8-10 cm per year and they attain a maximum size of approximately 180 cm (Harry *et al.* 2012). Females begin reproducing at 5 to 6 years off northern Australia and 7 to 8 years off the east coast of Queensland (Stevens and Wiley 1986, Harry *et al.* 2013). The maximum

recorded ages based on vertebrae are 8 to 13 years for males and 12 to 15 years for females (Davenport and Stevens, 1988; Harry *et al.* 2013). Vertebrae probably underestimate the maximum age of this species and tag returns indicate *C. tilstoni* is capable of living to at least 20 years (Stevens *et al.* 2000; Harry *et al.* 2013). Based on inshore fisheries catches all life history stages appear to occupy nearshore coastal habitats (Table 12.1) (Harry *et al.*, 2011). The life cycle of *C. tilstoni* is summarised in Figure 12.4.

Common blacktip shark

In Australia *C. limbatus* is born at approximately 72 cm and can attain a maximum size of 265 cm (Figure 12.3; Table 12.1) (Stevens, 1984; Macbeth *et al.*, 2009; Harry *et al.*, 2012). Off eastern Australia, males mature between 185 and 205 cm while females mature between 200 and 215 cm (Macbeth *et al.* 2009). Elsewhere, size at maturity varies between geographic regions with males maturing between 135-180 cm and females from 120-190 cm (Last and Stevens, 2009). Usual litter size is 4-7 (maximum 10) produced after a 10-12 month gestation. Individual females breed every other year, although a triennial reproductive cycle has been suggested in South Africa (Dudley and Cliff, 1993). Age at maturity in other parts of the world is 5-6 years for males and 6-7 years for females (Table 11.1) (Last and Stevens, 2009). Adult females are assumed to move in to coastal waters to give birth. Only neonates and juveniles of *C limbatus* are caught in east coast inshore fisheries in these habitats suggesting that adults generally prefer deeper water (Harry *et al.*, 2011). The life cycle of *C. limbatus* is summarised in Figure 12.5.

Distribution, habitat and environmental preferences

C. tilstoni is endemic to northern Australia and *C. limbatus* is found in subtropical and tropical waters worldwide. In Australia *C. limbatus* and *C. tilstoni* co-occur in subtropical and tropical waters however *C. tilstoni* are more common in tropical warmer waters and *C. limbatus* are more common in sub-tropical waters (Last and Stevens 2009; Ovenden *et al.* 2010) (Figure 12.5). *C. tilstoni* is found in continental shelf waters of tropical Australia and adults, juveniles and neonates appear to co-occur in coastal fishery areas (Harry *et al.*, 2011). The southern limits of its distribution are uncertain, as it has been confused with *C. limbatus*. On the east coast reported *C. tilstoni* has been reported as far south as Moreton Bay (27°S) based on vertebral counts (Harry *et al.* 2012), and as far south as Sydney (34°S) based on genetic samples (Boomer *et al.*, 2010). On the west coast, *C. tilstoni* is known to occur as far south as Dampier (21°S). *C. limbatus* adults appear to prefer deeper shelf waters since they are not generally encountered in the ECIFFF, while neonates and juveniles are found in shallow nearshore habitats (Harry *et al.*, 2011).
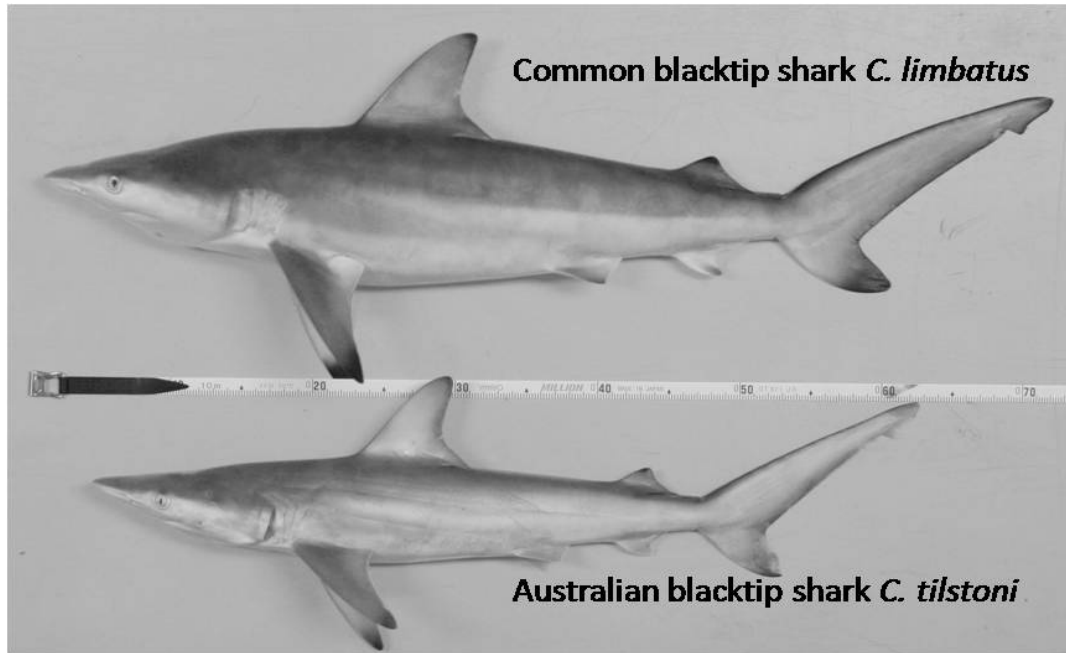
Figure 12.3. Comparative sizes of neonate blacktip sharks showing *C. limbatus* tending to be larger than *C. tilstoni* at birth (Source: Alastair Harry).

Table 12.1. Key aspects of the life history/ecology of *C. limbatus* and *C. tilstoni* that can assist in distinguishing between the two species. Sizes refer to SL = stretched total length. (Sources: Stevens, 1984; Davenport and Stevens, 1988; Last and Stevens, 2009; Macbeth et al., 2009; Ovenden et al., 2010; Harry et al., 2012; Harry et al., 2013).

Species	Australian distribution	Timing of birth	Mean size @ birth	Size @ maturity	Maximum size
C. limbatus	Most common in the sub-tropics	Oct-Jan Peak Nov	72 cm ±29sd	185-205 cm ♂ 200-215 cm ♀	265 cm
C. tilstoni	Most common in the tropics	Dec-Jan Region-specific	60-62 cm	105-120cm ♂ 120-125 cm ♀	180 cm

Predators and prey

The diets of blacktip sharks are known to contribute significantly to the natural mortality of valuable commercial prawns (Salini *et al.,* 1990; Brewer *et al*., 1991; Salini *et al.,* 1992). Stomach contents indicate that teleost fish are an important component of the diet of both species and there is some indication of a change in feeding depth with shark size (Stevens *et al.,* 1986).
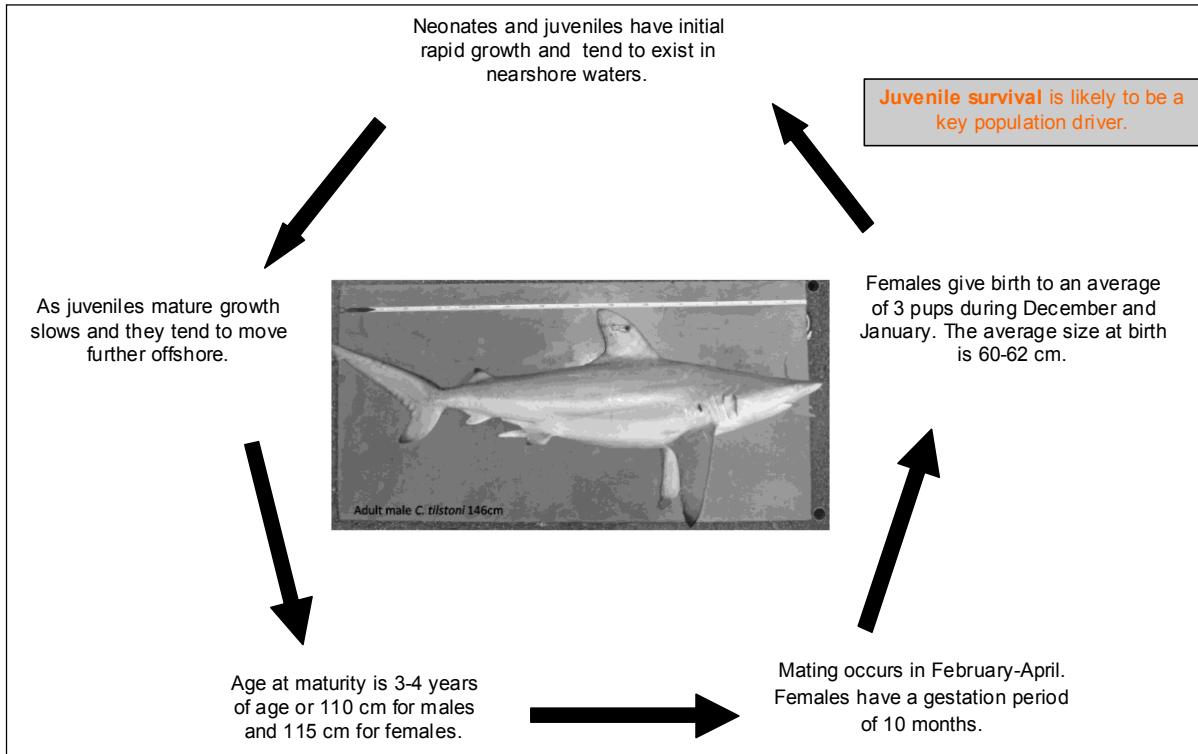
Figure 12.4. Summary of the life cycle of the Australian blacktip shark (*Carcharhinus tilstoni*).

Neonates and juveniles have initial rapid growth and tend to exist in nearshore waters.

Juvenile survival is likely to be a key population driver.

As juveniles mature growth slows and they tend to move further offshore.

Females give birth to an average of 3 pups during December and January. The average size at birth is 60-62 cm.

Adult male C. *tilstoni* 146cm

Age at maturity is 3-4 years of age or 110 cm for males and 115 cm for females.

Mating occurs in February-April. Females have a gestation period of 10 months.

Figure 12.5. Summary of the life cycle of the common blacktip shark (*Carcharhinus limbatus*).

Juveniles have initial rapid growth and tend to exist in near-shore waters.

Juvenile survival is a likely to be a key population driver.

As juveniles mature growth slows and they tend to move further off-shore. As adults they appear to pre-fer deeper waters.

Females give birth to 4-7 pups (max 10) in coastal waters between October and March. The average size at birth is 72 cm.

Mature as adults at 5-6 and 6-7 years for males and females respectively. Size at maturity is135-180 cm for males and 120-190 cm females. Maximum size is ~ 265 cm.

Females breed once every 2 years. Gestation is approximately 10-12 months.

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Recruitment

Given the low fecundity of blacktip sharks recruitment is likely to be heavily influenced by the abundance of mature females.
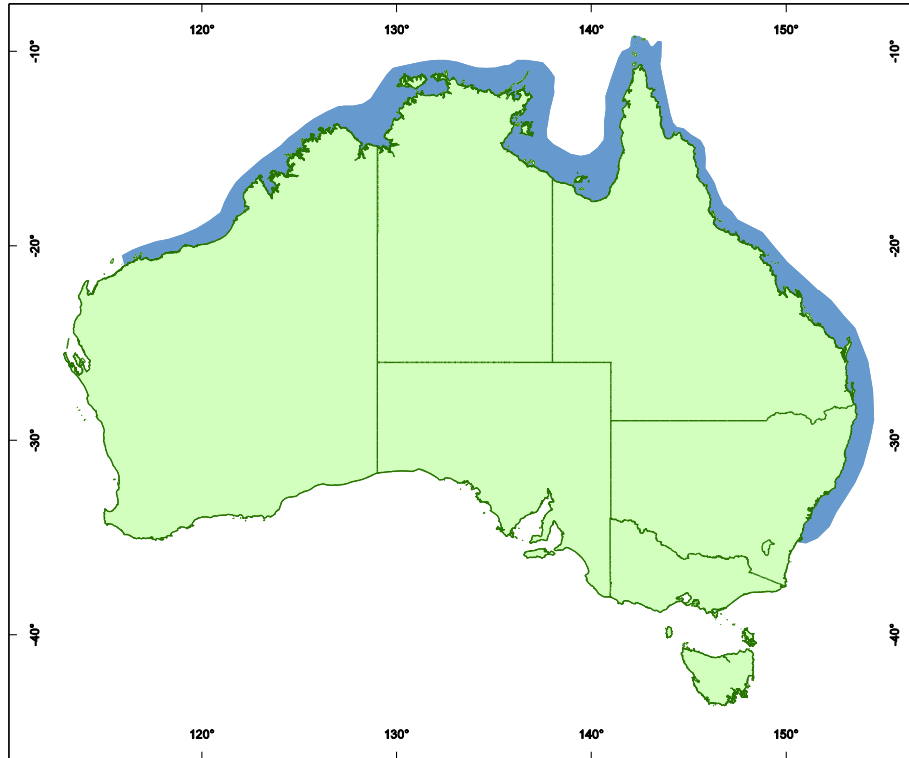
Figure 12.6. Distribution of *C. tilstoni* and *C. limbatus* within Australian waters.

Current impacts of climate change

Current impacts of climate change are unknown for the two blacktip species. A recent vulnerability assessment of sharks and rays on the Great Barrier Reef concluded that both the Australian blacktip and the common blacktip shark had a low vulnerability to climate change (Chin et al 2009).

Sensitivity to change

> **Key points:**
> - Sensitivity of blacktip sharks to environmental change is unknown although temperature may influence embryonic growth rate.

The sensitivity of each blacktip shark species to environmental changes is unknown. Although some coastal sharks in general show particular environmental preferences (eg. salinity: Heupel and Simpfendorfer, 2008; freshwater flow rates: Knip et al., 2011), they appear to be adapted to a range of environmental conditions including temperature, salinity and pH. Because sharks tend to give birth to relatively small numbers of young (or lay a small number of eggs) recruitment in shark populations is thought to be closely dependent on stock size and less affected by environmental conditions (Walker, 1998). However, Harry *et al.* (2013) noted a close correlation between ambient

environmental temperature and embryonic growth rate in *C. tilstoni* and spot-tail shark, *C. sorrah*. This observation suggests that species such as *C. tilstoni*, which spend their entire lives in the relatively dynamic coastal environments, may still be sensitive to environmental conditions.

Resilience to change

Key points:
- It is likely that both species are resilient to climate change given their high mobility and wide habitat/environmental preferences.

Globally, populations of *C. limbatus* are widespread covering a vast range in environmental conditions suggesting their high resilience to changes in the environment. Within Australia both black tip shark species occur over a relatively wide latitudinal range and environmental conditions and are therefore likely to be resilient to changes in their environment. They are also highly mobile animals enabling them to readily move between preferred environments. Blacktip sharks are therefore likely to be resilient to climate change.

Other

Key points:
- Future population levels of blacktip sharks will be influenced by prey availability and therefore impacts on fish species will affect sharks, depending on the species.
- The general low productivity of blacktip sharks, particularly *C. limbatus*, means they have a low capacity to recover from any future impacts of climate change.

Ecosystem level interactions
Sharks constitute a major fraction of the predator biomass in tropical waters (Blaber *et al.* 1989, 1990a; Salini *et al.* 1992) and as a consequence exert an important top down influence impact on tropical coastal ecosystems.

Additional (multiple) stressors
Sharks in general are vulnerable to overexploitation due to their slow growth, late maturity and low fecundity (Ovenden *et al.,* 2010). Currently shark fisheries in Australian waters are generally managed tightly and so increased pressure from any fishing sector in the future is unlikely. Despite this, fishing pressure may exacerbate any impacts on blacktip populations from climate-induced changes. Illegal, unregulated and unreported fishing has increased off northern Australia in recent years and could potentially affect these species (Field *et al.,* 2009).

Critical data gaps and level of uncertainty
Although there have been recent major advances in methods for distinguishing among the two blacktip species (Morgan et al., 2011; Harry et al., 2012), further information on the catch composition of each species is required to assess the impact of the ongoing targeting by commercial fisheries. Also, the recent evidence of widespread hybridisation between the two species suggests further research should investigate the fitness of hybrids (Morgan et al., 2012).

Acknowledgements

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13.   Coral trout, *Plectropomus spp.*

Author: David J. Welch

Common coral trout, *Plectropomus leopardus*. Photo: Anthony Roelofs.

Across northern Australia there are several species of the Family Serranidae that are known as coral trout. These belong to the genera *Plectropomus* and *Variola* and their relative abundance varies regionally and across the continental shelf. By number, the most common species in most regions of northern Australia is the common coral trout or leopard coral grouper (*Plectropomus leopardus*). Other species include bar-cheek coral trout (*P. maculatus*), bluespot coral trout (*P. laevis*), passionfruit coral trout (*P. areolatus*), highfin coral trout (*P. oligacanthus*), coronation coral trout (*Variola louti*) and lyretail coral trout (*V. albimarginata*) (Heupel et al, 2010). In fisheries across northern Australia *P. leopardus* comprises the vast majority of the total catch and so this review will primarily focus on this species.

The fisheries

- Most of the catch comes from the Great Barrier Reef on the Queensland east coast. Catches in Western Australia, Northern Territory and the Gulf of Carpentaria are negligible.
- Coral trout represent a valuable and important target species for all sectors in the Great Barrier Reef line fishery.
- Commercial catch is regulated by quota, while recreational catch is not well estimated.
- Coral trout are considered sustainably fished on the GBR.

Commercial catches of coral trout species are negligible in Western Australia and Northern Territory with most of the catch likely to be taken by the recreational fishing sector. In the Queensland Gulf of

Carpentaria line fishery the reported commercial catch is also very low with only 1.92 t taken annually in the period 2000 - 2009 (DEEDI, 2010). The charter fishing sector takes similarly low quantities, and assuming historically similar catch composition with the charter sector, the 2005 recreational coral trout catch in the Gulf of Carpentaria is likely to be approximately 14 t (DEEDI, 2010).

Queensland Reef Line fishery
Coral trout are the predominant target species for the Queensland Great Barrier Reef line fishery (RLF) historically comprising approximately 50% of the total catch (Welch et al, 2008). The fishery is multi-species, comprising in excess of 125 species, and multi-sectoral comprising commercial, recreational and charter fisheries. Fishing methods used are handlines (all sectors) and rod and reel (recreational and charter), with fishers operating from small vessels on individual coral reefs usually in depths less than 20 m (Welch et al, 2008). Since the mid-1990s, there has been a rapid growth of an export market for live fish, particularly coral trout (*Plectropomus spp*.), to south-east Asia (Mapstone et al., 2001; Sadovy et al., 2003), although a small number of vessels still supply dead product (Figure 13.1) (Welch et al 2008). The live fish market has increased the profitability of the RLF and the 2009-10 estimate of the commercial gross value of production of $45 million (DEEDI, 2011a) is based primarily on the coral trout catch component. Currently there are 369 commercial fishing endorsements for the RLF (RQ symbol) of which approximately 205 are active (DEEDI, 2011a).

Prior to 2004 the commercial sector was regulated mainly by effort controls, and the recreational and charter sectors had trip and/or bag (in possession) limits. For all sectors a minimum size limit (MSL) of 38 cm (TL; total length) was applied to coral trout for all sectors. In 2003–2004 management of the fishery changed substantially with the introduction of an annual total allowable commercial catch (TACC) allocated as individual transferable quotas (ITQs) for the key fishery species groups (coral trout, red throat emperor and 'Other' species). The coral trout TACC introduced was 1,350 t. Since quota was introduced the TACC has not been realised in any year and in 2009 the reported commercial harvest of coral trout was 1,028 t (80% of the TACC) (Figure 12.2) (DEEDI, 2011b). The MSL for *P. leopardus* and *P. maculatus* remained at 38 cm TL, and revised size limits for *P. laevis* were introduced with a minimum size of 50 cm TL and a maximum size of 80 cm TL for all sectors (Coral Reef Fin Fish Fishery Management Plan, 2003). There is also a seasonal spawning closure in place that prevents any fishing for coral reef finfish for five days around the new moon in October and November each year (DEEDI, 2011b). Other management arrangements include gear restrictions (eg. number of lines and hooks), boat size restrictions (max. 20 m for primary vessels), and restrictions on the number of fishing tenders for each licence. Extractive uses of the Great Barrier Reef Marine Park are also regulated through a zoning plan that includes extensive areas of no-take reefs.

Coral trout are also popular target species for recreational and charter fishing sectors. The harvest estimate for the charter sector in 2009/10 was 80 t while in the recreational sector for the years 1997, 1999, 2002 and 2005 harvest ranged from 196,000 – 332,000 fish. No estimate of recreational catch is available in terms of weight. The current stock status of coral trout in the GBR RLF is assessed as 'sustainably fished' (DEEDI, 2011b).
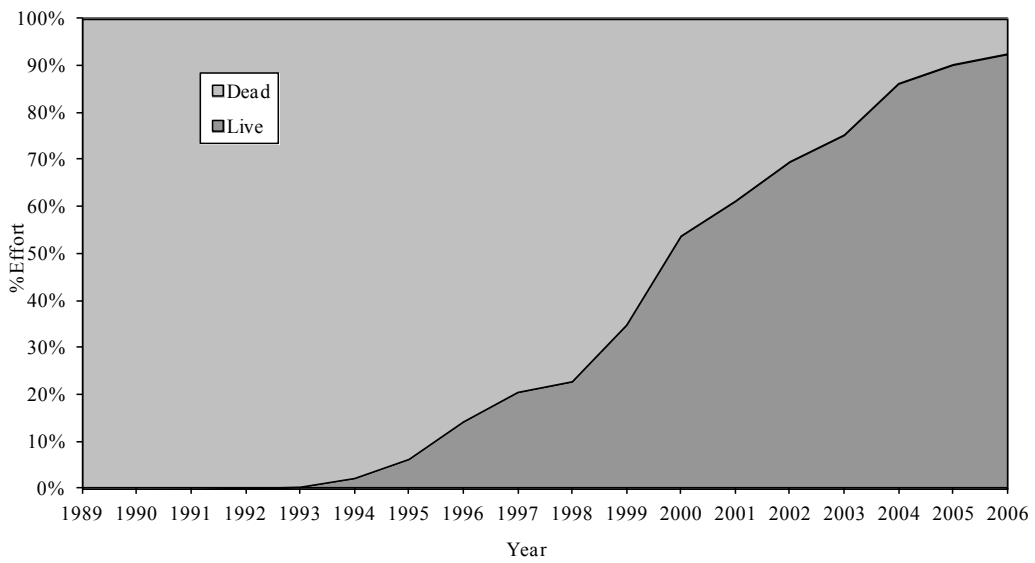
Figure 13.1. Changes in the commercial effort in the GBR line fishery showing the dramatic change in targeting dead to live product from 1989 to 2006. (Source: Welch et al, 2008).
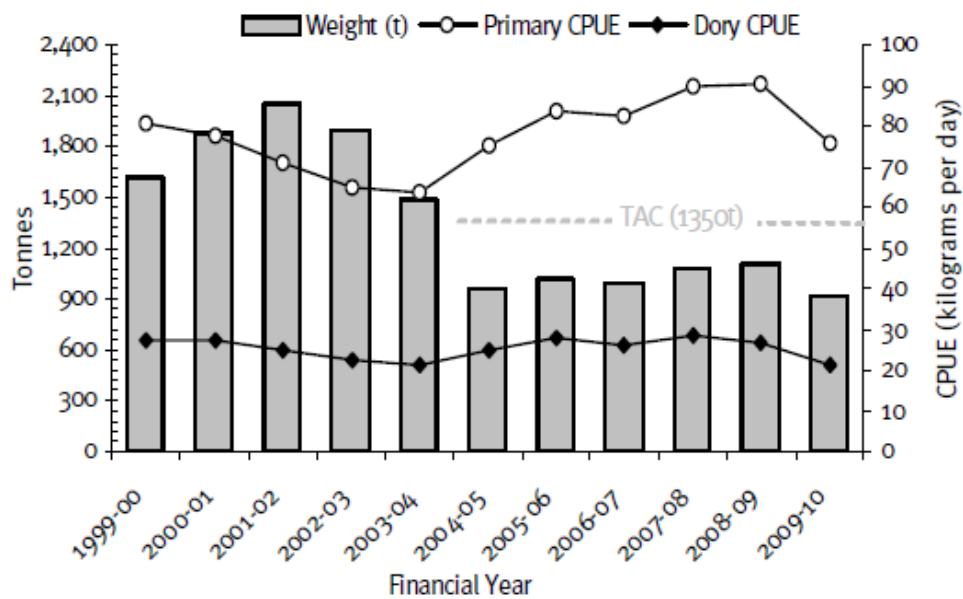
Figure 13.2. Commercial catch of coral trout from the Great Barrier Reef line fishery for the financial years (quota years) from 1999-00 to 2009-10. Catch-per-unit-effort (CPUE) for primary vessels and dories are also indicated. (Source: DEEDI, 2011b).

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Life history

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Key points:
- Coral trout are protogynous hermaphrodites; they mature first as females and change sex to become males as they get older and larger.
- *P. leopardus* are fast growing and early maturing and live for at least 17 years.
- Coral trout occupy a range of habitats but overall have a dependency on coral reefs and from a young age show strong site fidelity.

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Life cycle, age and growth

Coral trout are protogynous hermaphrodites meaning that they develop primarily as females and change sex during their life to become males (Goeden, 1978; Ferreira, 1995). Coral trout spawn either in pairs, small groups or large (>100 individuals) aggregations with peak spawning activity during new moons from September – December (Ferreira, 1995; Samoilys, 1997; Samoilys and Squire, 1994). The numbers of individual fish involved and the timing is variable among and within years but the onset of spawning appears to be correlated with rising sea water temperatures (> 24° on the GBR) (Samoilys, 1997). They are broadcast spawners that rush to the surface in pairs to release gametes into the water column (Samoilys and Squire, 1994).

Longevity in *P. leopardus* is at least 17 years (Lou et al, 2005) and growth was first estimated by Ferreira and Russ (1994) for the northern GBR. Growth is fast in the first 2-3 years and slows to an asymptote as they get older. Growth was described using the von Bertalanffy growth function with parameter estimates of L_∞ = 522 mm FL, K = 0.35, and t_o = -0.77. More recent VBGF parameter estimates covering a greater area of the GBR show regional variation for the respective parameters being[2]: L_∞ = 424 - 488 mm FL, K = 0.48 – 0.59 (Welch, 2001). *P. leopardus* can reach sizes in excess of 70 cm FL and 7 kg in weight.

Size and age at first reproduction in *P. leopardus* was first estimated to be 24 – 36 cm FL, and 2-4 years from samples collected in the northern GBR, while sex change can occur across a wide range of sizes and ages (Ferreira, 1995). Adams et al (2000) found that *P. leopardus* may exhibit regional variation in their reproductive strategies, particularly the size and age at which sex change occurs.

The inshore or bar-cheeked coral trout, *P. maculatus*, has similar growth characteristics to *P. leopardus* (Williams et al, 2008) and can reach at least 75 cm FL and 8 kg in weight. The oldest specimen examined by Ferreira and Russ (1992) was 12 years old however they are likely to have similar longevity as *P. leopardus*. *P. maculatus* also show similar reproductive strategies as *P. leopardus* with first maturity at ~ 30 cm and 2 years of age and sex change can occur across a wide range of sizes and ages (Ferreira, 1993). More recent work from the Torres Strait indicate *P. maculatus* are capable of reaching maturity (~ 25 cm FL) and changing sex at smaller sizes than *P. leopardus* (Williams et al, 2008).

The blue spot coral trout, *P. laevis*, grows substantially larger than other coral trout species and reaches sizes in excess of 120 cm FL and 25 kg. Despite this, longevity of *P. laevis* is probably similar

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[2] Estimates of t_o are not given because the fitting of growth models constrained t_o to a size at hatching of 1.62 mm (following Masuma et al, 1993; $t_o \approx 0$) to minimise biases from gear selectivity during sampling.

to that of *P. leopardus*, since Heupel et al (2010) sampled fish up to 14 years old but less than 100 cm FL. Spawning of *P. laevis* is also similar to *P. leopardus* but may extend farther into the Austral summer period (Heupel et al 2010). The length at 50 % maturity for females was estimated to be 45 cm FL however maturity can occur at 30 cm FL and 1 year old, while the length at 50 % sex change is 87 cm but can occur as small as 46 cm (Heupel et al, 2010).

Distribution, habitat and environmental preferences

Coral trout (*Plectropomus* spp.) are medium-large sized, relatively sedentary predatory species that prefer coral and/or rocky habitats and are distributed in tropical and sub-tropical regions including the eastern Indian Ocean and the western Pacific Ocean extending from Australia and Fiji to southern Japan (Randall and Hoese, 1986). Throughout northern Australia they can be found in a range of habitat types generally associated with reef habitat and from depths of 2 m to at least 40 m. Within Australia *P. leopardus* range from the Abrolhos Islands in Western Australia to SE Queensland in eastern Australia, with rare encounters south of these limits (Figure 13.4). *P. leopardus* are particularly common on the GBR and the Abrolhos Islands.

On the GBR *P. leopardus* are found across the continental shelf from inshore reefs and headlands to offshore barrier reefs, however they are most common on mid-shelf reefs (Newman et al, 1997). Other species of coral trout also show differential habitat/shelf preferences. The inshore (or bar-cheek) coral trout, *P. maculatus*, is so named for its preference of inshore reef areas while *P. laevis* is most often encountered on offshore reefs. *P. areolatus* are more commonly found in northern parts of the GBR and particularly the Torres Strait (Williams et al, 2008). The two *Variola* spp are far less common and tend to be sighted more often on offshore reefs, particularly on steep reef slopes.

In assessing microhabitat preferences for juvenile *P. maculatus*, although a range of different microhabitats were used, Wen et al (2012) found that approximately 60% of all fishes (127/212) preferred *Acropora* corals situated on loose substrates (e.g., sand), despite this specific microhabitat accounting for only 12.8% of benthic cover in the study areas. It is likely that other species of coral trout will have preferred micro-habitats during early and adult life history stages.
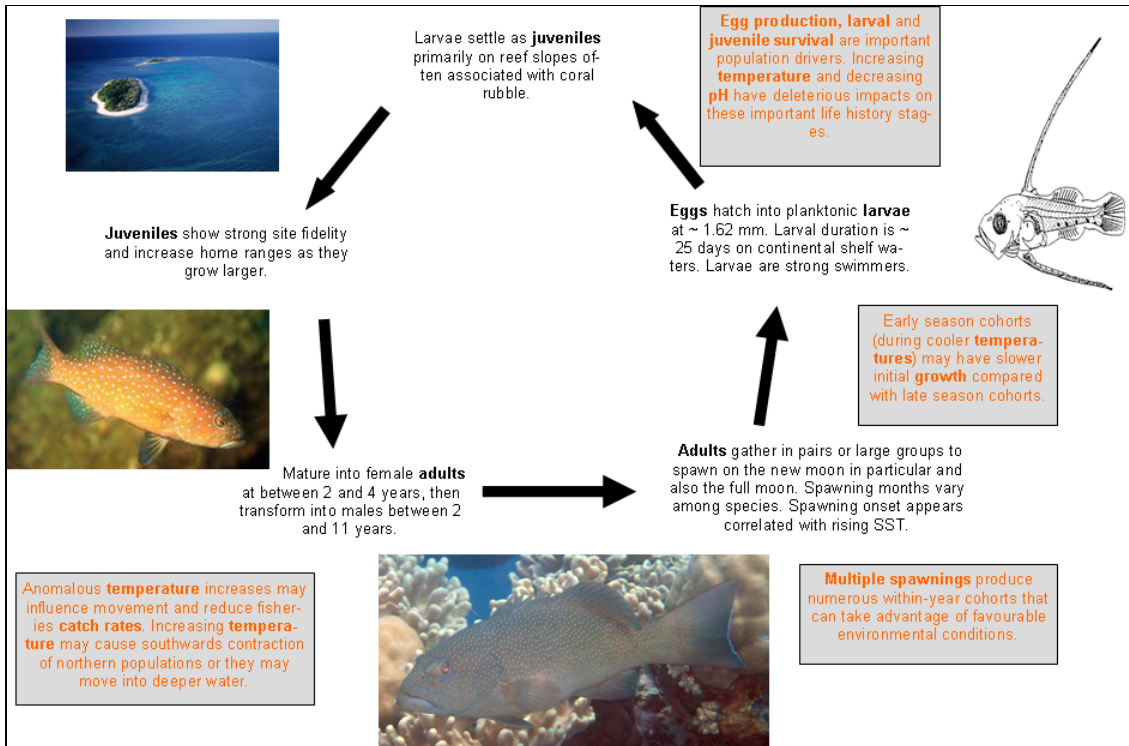
Figure 13.3. Generalised life cycle of the common coral trout, *P. leopardus*, and the stages of potential environmental impacts. Images: Masuma et al, 1993, GBRMPA, Anthony Roelofs.
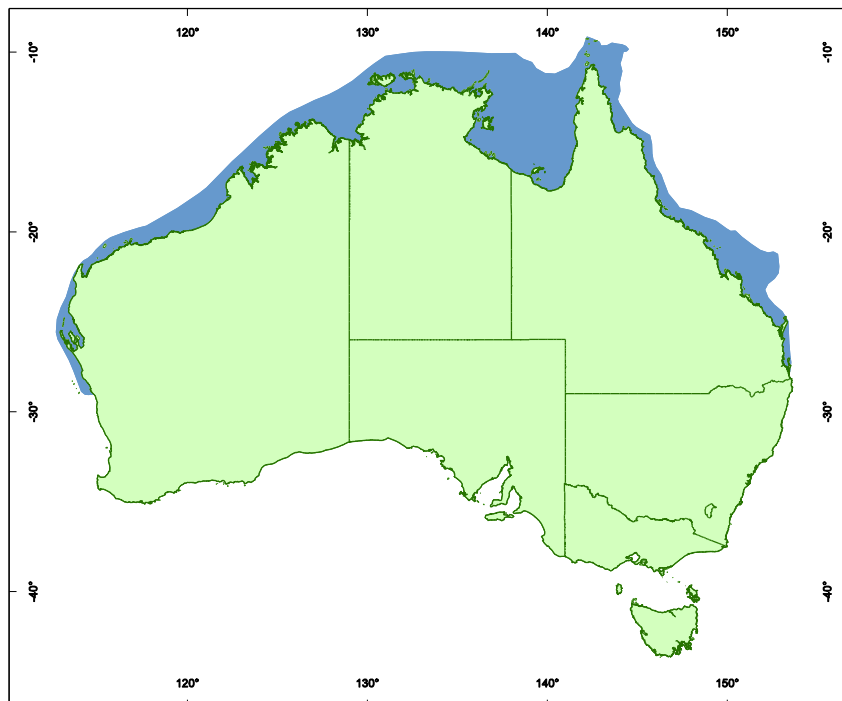

Figure 13.4. Distribution map showing the 'usual' range of the common coral trout, *Plectropomus leopardus,* within Australia.

Predators and prey

As juveniles coral trout consume a high proportion of benthic crustaceans, mostly penaeid shrimps, as well as small fish, whereas adults are almost entirely piscivorous (St John, 1999). Given the broad diet of *P. leopardus*, and because the two major prey families (Pomacentridae and Labridae) are diverse and abundant on coral reefs, St John et al (2001) concluded that coral trout are resilient to changes in abundances of particular prey species.

Recruitment

Coral trout eggs develop over a period of approximately 26 hours and have a size at hatching of 1.62 mm. The planktonic larval duration is approximately 25 days (Doherty, 1996; Masuma et al, 1993). Masuma et al (1993) give a very detailed account of the developmental larval stages. Once hatched, larvae show competent swimming capabilities with directional movement (Leis and Carson-Ewart, 1999). Variation in annual egg production, and in the survival of larval and juvenile stages, is significant and an important driver of population dynamics of coral trout with strong recruitment cohorts persisting over many years (Doherty and Williams, 1988; Doherty, 1996; Russ et al, 1996). Juvenile settlement occurs on reefs, primarily on reef slopes deeper than 4 m and they show a strong preference for habitat with a high proportion of coral rubble, algae, sand and rock. Recruits also show strong site fidelity and increase their home range size as they grow in size (Light, 1995). Light (1995) also presented evidence that earlier cohorts in a season, when temperatures are lower, exhibit slower initial growth compared with later cohorts when temperature is higher.

Current impacts of climate change

> **Key points:**
> - Cyclones have been shown to affect fishery catch rates of coral trout long after (many months) the passing of the cyclone with social and economic impacts on fishers.

Research by Tobin et al. (2010) demonstrated depressed catch rates of coral trout in the RLF following the crossing of Cyclone Hamish in 2009, a very large (Category 5) tropical cyclone, over parts of the Great Barrier Reef. The impact on the fishery was environmental, social and economical as some boats had to move substantial distances to other ports to remain profitable, or remain in their home port resulting in loss of fishing crew. The shift to other regions also caused localised stock depletions in some areas and increased conflict regarding resource use. The research also examined Cyclone Justin (1997) which, although a less intense system was a long-lived cyclone, and resulted in significant decreases in coral trout catch rates accompanied by significant increases in catch rates of red throat emperor (*Lethrinus miniatus*), the secondary target species of the RLF. Underwater visual surveys conducted following Cyclone Hamish documented structural reef damage as high as 66 % on some reefs, however the same surveys also observed nominal increases in coral trout abundances. There was no apparent correlation between sea surface temperature and catch rate. For Cyclone Justin however, a distinct cool water anomaly was found to be the most likely driver of decreased coral trout catch rates (reduced by up to ~50 %) and increased red throat emperor catch rates (increased by up to ~200%) (Tobin et al 2010). The impacts were spatially and temporally variable for each cyclone making general statements about likely impacts of cyclones highly uncertain.

There are also several recent anecdotal reports that describe an increase in the sightings and captures of coral trout in the SE Queensland region, relative to historical levels.

Sensitivity to change

- Increases in temperature above ~28°C will have deleterious effects on coral trout early life history stages.
- Water chemistry changes may impact juvenile survival by altering predator-prey interactions.
- Cool water anomalies and/or cyclones may depress coral trout fishery catch rates.

Recent experimental studies on the effects of temperature and water chemistry on *P. leopardus* have greatly advanced our knowledge of the sensitivity of coral trout, and potentially other coral reef fish. Under different temperature regimes ranging from 24°C to 33°C, Pratchett et al. (2013) found that survival of larval coral trout is significantly reduced at increased temperatures above 28°C over the endogenous nutrition phase. The endogenous nutrition phase is the period between developing embryo and first feeding on live prey items (exogenous feeding). The study also found that at higher temperatures larvae had smaller initial yolk reserves, increased metabolic rate, were significantly smaller at the end of the endogenous phase, and had a more restrictive diet due to a smaller mouth gape, explaining the higher mortality observed. Further, at higher temperatures the duration of coral trout sperm motility was decreased, egg hatching rate was lower, and egg development showed increased irregularities. pH did not appear to have any impact on egg development and survival (Pratchett et al, 2013).

Pratchett et al (2013) found no difference in thermal sensitivity between northern and southern coral trout populations (separated by > 1200 km). The implications of this are that northern populations are likely to express responses to warming waters before southern populations. This could be a contraction of the species range southwards or redistribution of animals to deeper waters. At temperatures greater than 30°C the energy demands on coral trout became so great that normal function is likely to be compromised (Pratchett et al., 2013).

A related experimental study found that juvenile *P. leopardus* are sensitive to changes in water chemistry. At elevated pH levels juvenile coral trout became more attracted to the odour of predators and were more significantly more active and more inclined to move away from shelter, making them more vulnerable to predation. Munday et al. (2012) reared juvenile coral trout in laboratory conditions under different levels of pCO_2 (~495, 570, 700 and 960 µatm). The results showed that above 600 µatm CO_2 fish were more active and ventured further from shelter, and actually were attracted to the odour of predators. Similar research on a larval coral reef fish (*Amphiprion percula*) also showed a breakdown in the olfactory abilities in detecting predators with changes in water pH (Dixson et al., 2010).

The only other documented evidence of environmental effects on coral trout appear to be the influence of cyclones and temperature on catch rates, whereby cooler water can reduce catch rates (Tobin et al, 2010). Such water incursions can also be induced through upwelling and changes in water current patterns and pathways on the GBR are poorly understood. Other sources of evidence are either anecdotal or on similar species. For example, based on fishers' reports, coral trout may be moving farther south on the east coast of Queensland.

Research on other species show differential effects of changes in temperature and water chemistry on aspects of the species life history. In one example on a species related to coral trout, *Epinephelus malabaricus*, Yoseda et al (2006) found that the mean volume of yolk sac at larval onset of mouth opening and at onset of feeding was significantly larger at lower temperatures (25 °C) compared with higher temperatures (28 °C and 31 °C). They also found that larvae tended to absorb the yolk sac and consume the oil globule more rapidly with increasing temperature.

General conclusions have also been made about coral reef fish stating that warmer water temperatures are likely to increase larval development thereby reducing the planktonic larval stage, which in turn will reduce dispersal capabilities and alter spatial scales of connectivity (Munday et al, 2009).

Resilience to change

Key points:
- Coral trout show plasticity in their life history stages and have a broad diet making them resilient to changes in local conditions.
- Although they have a moderately wide thermal tolerance range, northern areas of their range will be approaching the maximum for normal function (~30° C) in the medium term future.

Coral trout have been shown to have variable growth rates, as well as size and age at maturity and sex change depending on location and possibly population densities (Adams et al, 2000; Welch, 2001) indicating they can adapt to changing environmental and population conditions. Coral trout (*P. leopardus*) also have a broad diet with two of their major prey families (Pomacentridae and Labridae) among the most diverse and abundant on fish families on coral reefs suggesting coral trout are resilient to changes in abundances of particular prey species (St John et al, 2001). The thermal tolerances of coral trout would appear to have an upper threshold of approximately 30° C (Pratchett et al., 2013). This corresponds with known distributions for *P. leopardus*, *P. maculatus* and *P. laevis* which occur across a range of latitudes with water temperature ranges from approximately 22 – 30° C, suggesting a moderately wide temperature tolerance. Coral trout also use rising sea water temperatures as a cue for spawning (> 24°C on the GBR) (Samoilys, 1997) and so under climate change scenarios of increasing water temperatures are likely to avoid the critical thermal thresholds that negatively affect larval development described above resulting in earlier spawning.

Other

Key points:
- Future impacts on coral reef habitats will also impact coral trout populations.
- Better understanding of the ability of coral trout to adapt to increases in temperature and acidification are required, as are better estimates of recreational harvest.

Ecosystem level interactions
Coral trout species are one of the most abundant coral reef fish predators, particularly on the GBR. They are therefore likely to be an important functional group in the functioning of coral reef ecosystems. The interactive effects of competition and predation, particularly during early life history stages, under a changing climate are poorly understood.

Additional (multiple) stressors

Coral trout appear to have preferred micro-habitat, particularly at the juvenile stage, which may be important for early survival (Wen et al., 2012). The predicted climate change impacts on coral reef habitats (Bell et al., 2011; Pratchett et al., 2011) could therefore indirectly influence coral trout population replenishment and exacerbate the effects of more direct impacts such as temperature and water chemistry. Fishing is the major potential stressor on coral trout populations however, current management of coral trout stocks in Australia is considered to be robust and stocks considered to be sustainably fished at current levels. However, recreational catch is expected to increase as human population increases, thereby intensifying the pressure on target fish stocks.

Critical data gaps and level of uncertainty

Critical information needs are the effects of temperature and pH on the different coral trout life history stages. Research into the adaptive capacity of coral trout to predicted changes in temperature and pH would also help put current knowledge in perspective. Coral trout are a popular target species by recreational anglers across northern Australia and currently estimates of the harvest by this sector are poor. More robust estimates of recreational catch are needed.

Acknowledgements

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14. Dusky Flathead, *Platycephalus fuscus*

Authors: Richard J. Saunders and David J. Welch

The Dusky Flathead, *Platycephalus fuscus*, is a member of the family Platycephalidae (the flathead) of the order Scorpaeniformes. The species is restricted to the east-coast of Australia from eastern Victoria to north Queensland and is a very important recreational species in NSW and Qld
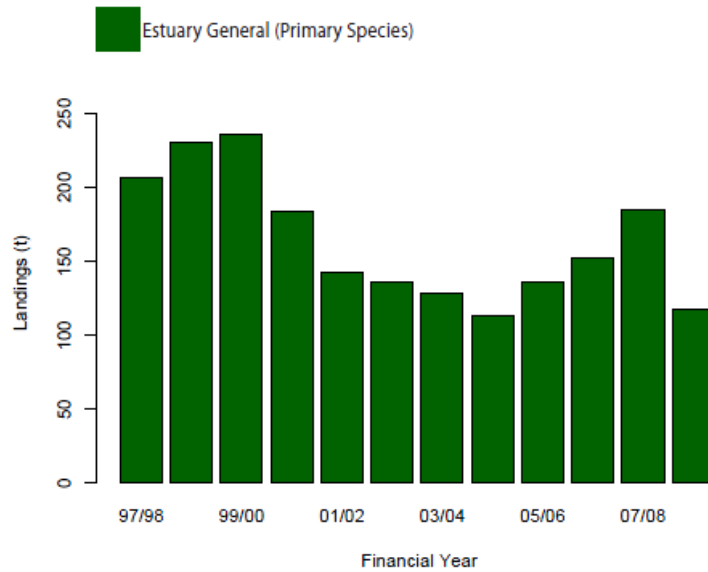
The fisheries

- Commercial catch is taken in inshore net fisheries in Qld and NSW.
- Recreational catch is much larger than the commercial catch.

Queensland

The dusky flathead is landed by commercial fishers in the East Coast Inshore Fin Fish Fishery (ECIFFF) and comprises <1 % of the total species composition by weight (Simpfendorfer et al., 2007). In 2009-10 there was 57 t of flathead reported in the commercial sector and the annual average over the past four years is 66 t. In the tropics dusky flathead form the bulk of flathead catches. Given the size limits in place for this species the fishery harvests predominantly females. Catches and catch rates have been stable and the species is currently considered to be sustainably fished (DEEDI, 2011a). RFISH diary surveys done to assess recreational catch in Queensland for the years 1997, 1999, 2002, and 2005 record catch of flathead ranging from 133 t in 1999 to 70 t in 2005 (McInnes, 2008). A large proportion of this catch is likely to be dusky flathead but the species composition is unknown. The species has minimum and maximum size limit and bag limits are in place for the recreational sector.

New South Wales

Commercial landings of dusky flathead are restricted to the Estuary General Fishery in NSW (Rowling et al. 2010) and are higher than the Qld commercial sector. Commercial catches in NSW since 1997/98 have generally been in the range of approximately 120 - 230 t. Commercial catches are mainly comprised of female fish (Gray et al, 2002). The catch in this fishery has varied between ~120 – 180 t in recent years after a large drop following a buy-out of many commercial fishers during 2000 (Figure 14.1). Historically, commercial catch of dusky flathead since 1952/53 has generally remained between 150 and 250 t per annum (Rowling et al. 2010).

Figure 14.1. Commercial catch of dusky flathead in NSW commercial fisheries from 1997/1998 to 2008/09 (figure extracted from Rowling et al. 2010).

The commercial catch of dusky flathead in NSW is dwarfed by the recreational catch which is thought to lie between 570 and 830 t (Rowling et al. 2011). Henry & Lyle (2003) report flathead to be the second most prominent group taken by recreational fishers in Australia, however, no analysis of flathead catch by species was done. Dusky flathead are assessed as fully fished in NSW waters. They have a minimum size limit only but a restriction of one fish > 70 cm TL, with a recreational bag limit of 10 (Rowling et al. 2011).

Life history

- Commercial catch is taken in inshore net fisheries in Qld and NSW.
- Recreational catch is much larger than the commercial catch.

Life cycle, age and growth

Dusky flathead are the largest flathead species attaining 1.2 m SL and 15 kg (Gomon et al. 2008). Growth has been well described by Gray & Barnes (2008) for NSW but several studies across the range of species have also considered age and growth (e.g. Dredge 1976; West 1993; Gray et al. 2002). Gray & Barnes (2008) reported sexually dimorphic growth for dusky flathead. The von Bertalanffy growth parameters for females were: L_∞ = 127.59 mm, K= 0.084, t_0= -2.39 and for males: L_∞ = 43.21 mm, K= 0.714, t_0= -0.67 (Gray & Barnes 2008). There is some evidence that dusky flathead from Victoria attain sexual maturity at a smaller size than those from southern Queensland (*see* Kailola et al. 1993).

Spawning occurs in northern Queensland from September to March (Dredge 1976), in Moreton Bay from November to February, and January to March in NSW and Victoria (Kailola et al. 1993). These are all periods associated with an increase in day length and water temperature (Dredge 1976). The species is likely to be multiple batch spawner and has high fecundity producing between 294,000

and 3,948,000 pelagic eggs (Gray & Barnes 2008). The larvae are pelagic and are described in Neira et al. (1998).

There has been some speculation that dusky flathead are protandrous hermaphrodites (Dredge 1976, Kailola et al. 1993) but this was based on observations of sized based sex ratios and no histological or physiological studies have found evidence for this. It is now considered more likely that the species exhibits dimorphic growth between the sexes resulting in exclusively large females and a very high proportion of males in the smaller size classes (Gray & Barnes 2008).

This species was successfully reared under laboratory conditions for a pilot program of stock enhancement in south-east Queensland (Butcher et al. 2000; 2003). Eggs were developed successfully at 23°C, and as 12 mm hatchlings they were transferred to 24–26.5°C ponds prior to being released at 35–50 mm (Butcher et al. 2000; 2003). The life cycle of dusky flathead is presented in Figure 14.2 with comments on potential influences of environmental variables on the different life history stages.
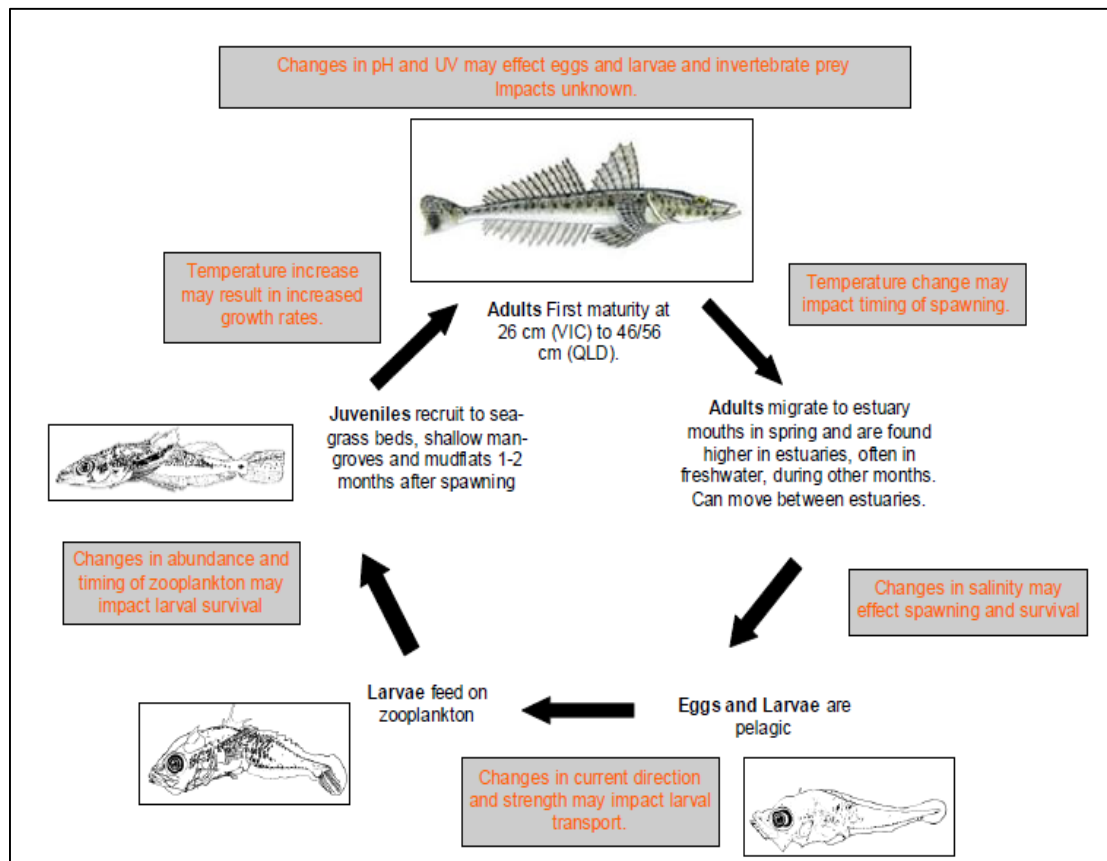
Figure 14.2. Generalised life cycle of the dusky flathead, *P. fuscus*, and the stages of potential environmental driver impacts (Source: Hutchinson, 2011).

Distribution, habitat and environmental preferences

The species is restricted to the east-coast of Australia from eastern Victoria to north Queensland (Figure 14.3). Dusky flathead occur in inshore coastal and estuarine environments usually associated with soft substrates, including mud, sand and seagrass. Movement studies show that dusky flathead are capable of moving long distances within an estuary (> 30km) and moving between estuaries (West 1993; Hindell 2008).
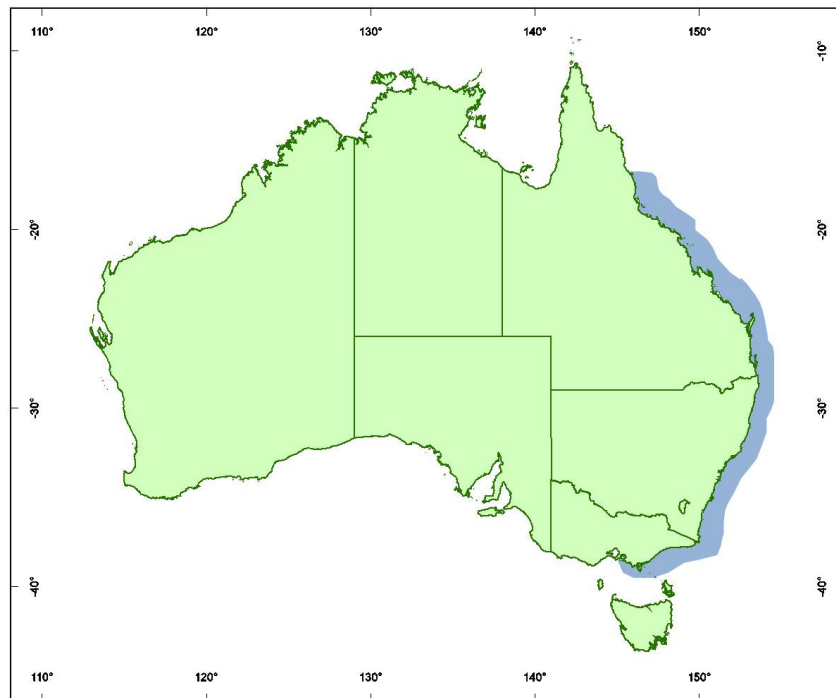
Figure 14.3. Distribution of dusky flathead.

Predators and prey

Flathead have been recorded in the diet of dolphins (Parra & Jedensjö 2009) and elasmobranchs (Walker 1989; Braccini et al. 2005; Treloar et al. 2007). As larvae they are likely to be taken by a wide range of teleosts. Dusky flatheads are primarily ambush predators (Dredge 1976; Kailola et al. 1993). The diet includes include fish, crustaceans, molluscs and polychaetes (Dredge 1976).

Recruitment

Recruitment processes of dusky flathead are not well understood. Larvae have been captured between September and May in estuaries and coastal waters of New South Wales (Gray and Miskiewicz 2000) and juveniles recruit to bays 1–2 months after spawning (Hindell 2008). Age structures of dusky flathead collected from commercial catch samples from four different estuaries of NSW collected over 2-3 years suggested that inter-annual recruitment can be highly variable (Gray et al., 2002).

Current impacts of climate change

There are no known current impacts of climate change on dusky flathead.

Sensitivity to change

- Rainfall has a positive correlation with dusky flathead catch and cpue
- Very little else is known of the sensitivity of dusky flathead to environmental variation

From a study in the Logan River, Queensland, the total catch of estuarine species in fisheries catch was been shown to be linked to the amount of freshwater runoff, particularly for flathead species (Loneragan and Bunn, 1999). A more recent study in different regions of the Queensland east coast found a significant positive correlation between annual coastal rainfall, the Southern Oscillation Index (SOI) and fisheries catch and cpue of flathead (Meynecke et al., 2006).

Resilience to change

- Given their latitudinal range they are likely to have a wide thermal tolerance, although this could be moderated if local stocks exist.

The latitudinal range of dusky flathead along almost the entire east covers a wide range in water temperatures and suggests a wide thermal tolerance for dusky flathead. It is not known, however, whether the east coast is comprised of a single stock or separate stocks. A study of dusky flathead commercial catches from four estuaries from different regions of NSW indicated differences in age, size and sex structures of the catch along with differences in mean-size-at-age suggesting the possibility of separate stocks (Gray et al, 2002). The fewer the number of stocks the more resilient dusky flathead are likely to be to changes in environmental conditions. Dusky flathead are considered to be generalist predators with a range of prey species making them resilient to changes in the availability of prey species.

Other

- Current harvest levels of dusky flathead could be nearing over-exploitation status.
- Recreational catch levels are a critical area for future assessment, as is the sensitivity of early life history stages to environmental changes.

Ecosystem level interactions

Ecosystem scale interactions are not well understood as with all species. Changes in the community structure of plankton have already been documented to have occurred in response to climate change. Jordan (1998) linked strong year class strength of southern sand flathead with peaks in the abundance of plankton. It is therefore likely that changes in the plankton will influence dusky flathead populations.

Additional (multiple) stressors

Fishing effort for dusky flathead is high in both NSW and Qld, particularly by the recreational sector. Although stock status in each state is considered to be 'sustainably fished', estimates of total mortality are considered to be high suggesting stocks may be subject to over-exploitation (Gray et al., 2002). Habitat impacts of climate change may affect all flathead species since they are benthic preferring soft substrates. Being an inshore and estuarine species dusky flathead will also be exposed to land-based impacts such as changes in water quality and salinity, and they are known to absorb a wide range of pollutants (Mondon et al., 2001).

Critical data gaps and level of uncertainty

One of the major information gaps for dusky flathead is knowledge of the sensitivity of each life history stage to changes in particular environmental variables such as temperature, salinity, pH, rainfall and extreme events. The larval and juvenile stages are potentially the most sensitive. Currently recreational harvest of dusky flathead is high and will only increase as human populations increase. Better estimates of recreational harvest levels are required to better manage the potential for cumulative impacts resulting in over-exploitation.

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15.   Golden snapper, *Lutjanus johnii*

Authors: Thor Saunders and Emily Lawson

A golden snapper taken in the Northern Territory. Photo courtesy of Jenny Ovenden.

The fishery

> **Key points:**
> - The Northern Territory Coastal Line fishery is the only northern Australian fishery that takes significant quantities of golden snapper.
> - Golden snapper are predominantly targeted by recreational fishers across their range.

Western Australia

A very small amount of golden snapper (2 t) was reported from Western Australia commercial fisheries in 2010. The recreational and Indigenous catch of this species in WA is unknown (Department of Fisheries 2011).

Northern Territory

The Coastal Line Fishery operates in the near-shore waters of the Northern Territory (NT) and primarily targets golden snapper (*Lutjanus johnii*) and black jewfish (*Protonibea diacanthus*) using hook and line gear. The fishery comprises commercial, recreational, charter and Indigenous sectors and there is considerable overlap in the range of species harvested. The Department of Resources (DoR), in consultation with the Coastal Line Fishery Management Advisory Committee (CLFMAC), is currently reviewing the management arrangements for the fishery to maintain the sustainable harvest of coastal fish species by all sectors. In 2010, 5 t of golden snapper was caught by the commercial sector (Figure 15.1). This species has been less targeted in recent years as operators have been able to get a better price for black jewfish (Northern Territory Government 2011).

The National Recreational and Indigenous Fishing Survey (NRIFS) conducted in 2000-01 indicated that of the ~600,000 fish harvested (i.e. caught and kept) by recreational fishers in the NT, the most common were snappers (23% of the total harvest). Golden snapper accounted for the largest portion of the snapper harvest being estimated at 68,000 fish (Coleman 2004). Fishing Tour Operators (FTOs) caught 15,382 golden snapper in 2010. Of these, 53% were released. Golden snapper harvest by the Indigenous sector is considered to be low due to the locations generally thought to be targeted by this sector (Northern Territory Government 2011).

Queensland

Golden snapper are harvested in both the Gulf of Carpentaria Inshore Fin Fish Fishery (GOCIFFF) and the East Coast Inshore Fin Fish Fishery (ECIFFF) however are not reported separately due to the very low (< 2 t) annual catch. Golden snapper are an important recreational species in Queensland, but they have not been reported as a separate species during recreational fishing surveys (DEEDI 2010a&b). Much of the targeting of golden snapper in Queensland has historically occurred in estuaries where juveniles are found, however in recent years increased targeting of larger adults on nearshore reefs and headlands has occurred due largely to the introduction of more efficient methods (eg. soft plastics) and advances in technology. The Indigenous catch of golden snapper in Queensland is unknown.
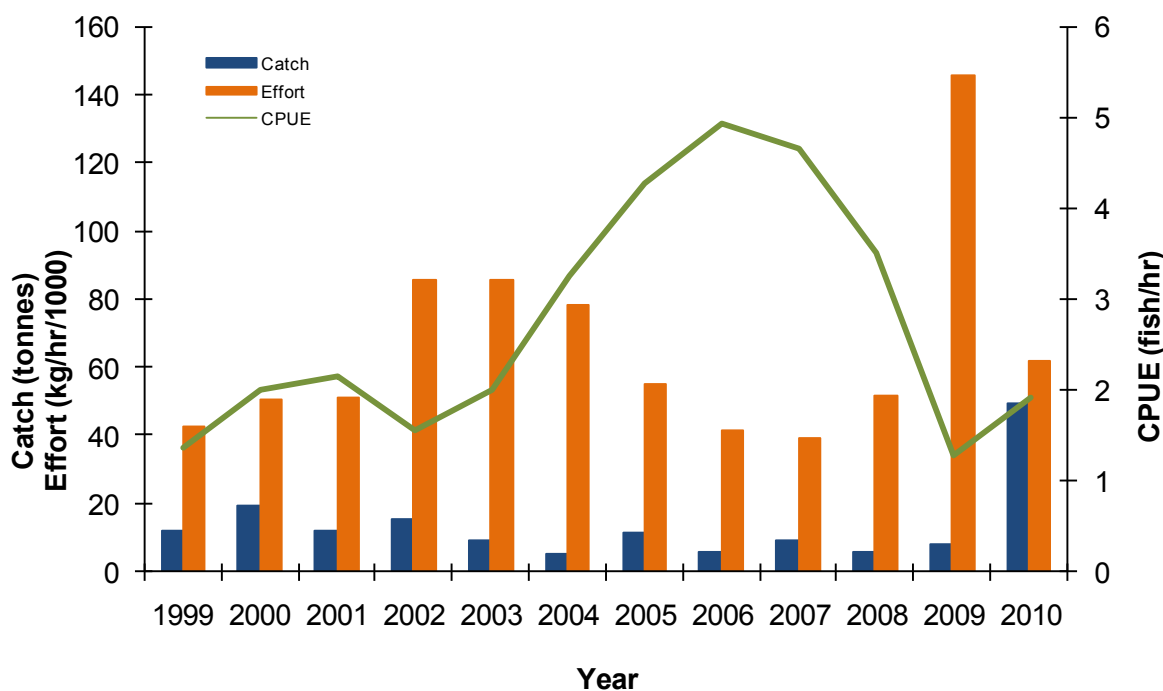

Figure 15.1. Annual catch, effort and CPUE for golden snapper in the NT Coastal Line Fishery between 1999-2010.

Life history

<div>

Key points:
- Golden snapper are slow growing and late maturing making them prone to overfishing.
- Juveniles are found in nearshore embayments and estuaries and remain in this habitat until maturity whereupon they move to nearshore reefs and headlands.

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Life cycle age and growth

Golden snapper are gonochoristic (i.e. separate sexes throughout life) and can grow to at least 90 cm and 12.4 kg and live up to 20 years of age (Marriot and Cappo 2000). Despite growing reasonably quickly as juveniles (up to 30cm/year), growth slows substantially upon reaching maturity (Hay *et al.* 2005). Growth rate has also been shown to vary with latitude with quickest growth occurring at southern latitudes in the tropics (Starling and Cappo 1996). The onset of first maturity in golden snapper is also related to age, not size as faster growing southern QLD fish reach maturity at about the same age as slower growing northern QLD fish (Northern Territory Government unpublished data). Maturity for this species is reached at 63 cm or eight years of age for females and at 47cm or five years of age for males (Hay *et al.* 2005).

In the Northern Territory this species undergoes a prolonged spawning period from early September to late April (Hay *et al.* 2005). From the aquaculture experience with golden snapper, large females (72-75 cm TL) can spawn at least 2.83 million eggs over the course of four consecutive nights (Lim *et al.* 1985) suggesting that this species is quite fecund. There is also a moderate, positive, linear correlation between fish size and total egg production (Northern Territory Government unpublished data). Once hatched, golden snapper larvae are typical of most reef fishes and enter the nearshore environment to settle to benthic substrates. As larvae grow into juveniles they move to estuarine habitat where they remain until maturity whereupon they move to deeper, nearshore waters (Starling and Cappo 1996, Kiso and Mahyam 2003). Trapping surveys suggest that golden snapper are most active at night (Travers *et al.* 2006). Golden snapper have also been shown to suffer from barotrauma related mortality when they are caught from waters deeper than 15m (Northern Territory Government 2011).
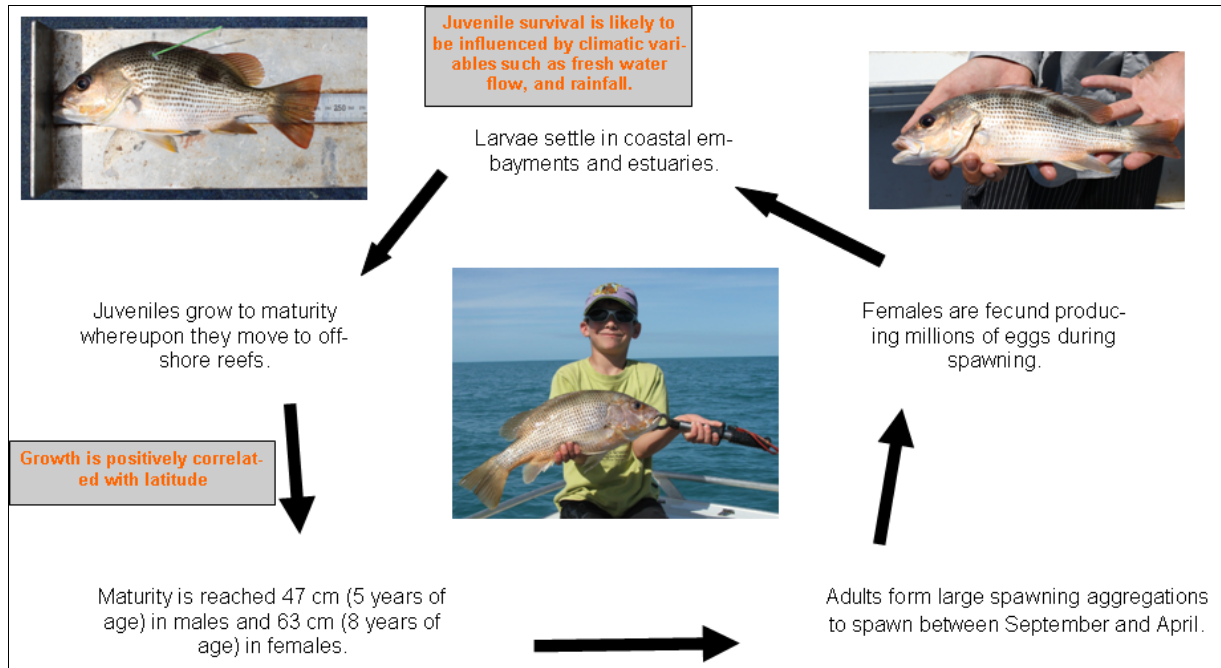
Figure 15.2. Summary of the life cycle of golden snapper, and the points of exposure to relevant climate change drivers.

Distribution, habitat and environmental preferences

Golden snapper have a wide geographic range throughout the Indo-West Pacific, inhabiting tropical inshore waters from East Africa to Fiji and northern Australia to just south of Japan (Hay *et al* 2005). In Australia they are distributed from the Kimberley region (~124°E) in north-western Australia, across northern Australia and extend down the east coast to at least 14°S (Anderson & Allen, 2001, Travers *et al* 2006, Hoese et al. 2007) and tagging studies have shown that this species is distributed to 24°S on the east coast (Bill Sawynok, unpublished data; Figure 15.3).

Their preferred habitat in deep and shallow water is around reefs, rocks, snags and pinnacles (Hay *et al* 2005, Travers *et al* 2010) but often move out onto adjacent sand areas possibly to feed (D. Welch pers. obs.). Juveniles are more regularly encountered in creek systems and mangroves, whereas the larger adult fish are encountered on coastal and nearshore reefs (Hay *et al.* 2005, Kiso and Mahyam 2003).
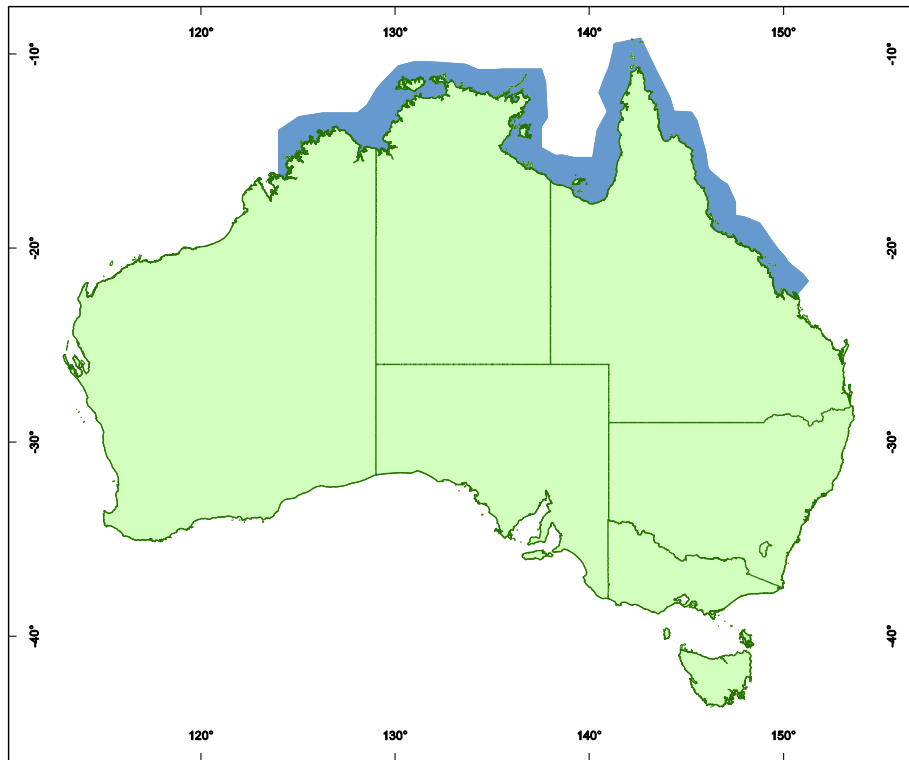
Figure 15.3. Australian distribution of golden snapper.

Predators and prey

Lutjanids are active predators feeding mainly at night on a variety of items, but fishes are dominant in the diet. Other common foods include crabs, shrimps, various other crustaceans, gastropods, cephalopods, and planktonic organisms (Randall *et al.* 1996, Travers *et al.* 2010). Juvenile golden snapper in estuaries feed on small crustaceans, and they shift their preference as size increases (Kiso and Mahyam 2003) and as adults prey mainly on fish and larger crustaceans (Druzhinin 1970). Larger fish and sharks are likely predators of golden snapper particularly as juveniles when they inhabit estuarine habitat.

Recruitment

Very little is known about the recruitment dynamics of golden snapper. Their larvae enter coastal embayments and estuaries so survival is likely to be influenced by ocean current strength and direction, river flow and rainfall and water temperature, salinity and pH.

Current impacts of climate change

Key points:
- There is recent evidence to suggest a southern expansion of this species occurring on the east coast.

Current impacts of climate change are largely unknown for this species. However, there may have been a recent southern expansion on the east coast of Australia. This evidence is based around tag

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recapture data over 10+ years that has indicated a higher abundance of this species a degree of latitude further south than when the study was initiated (Bill Sawynok, unpublished data).

Sensitivity to change

> **Key points:**
> • The sensitivity of golden snapper to environmental changes is unknown.

The sensitivity of golden snapper to environmental changes is unknown. Like other species with their entore life history occurring in the estuarine and nearshore environment, it is highly likely that golden snapper populations are strongly influenced by annual rainfall and river flow regimes.

Resilience to change

Although golden snapper occupy inshore habitats that are prone to high annual variability in environmental conditions, their resilience to climate-related changes are unknown.

Other

> **Key points:**
> • Information on the influence of changes in environmental variables on golden snapper life history stages is lacking.
> • Better estimates of recreational harvest as well as the size characteriestics of the catch are needed to better quantify fishing impacts.

Ecosystem level interactions
The influence that fluctuations in golden snapper abundance on the ecosystem they inhabit are unknown. However, any impact is unlikely to be significant as there are a variety of other closely related species that utilise tropical coastal reefs that could readily take their place in this ecosystem.

Additional (multiple) stressors
Golden snapper are heavily targeted by recreational anglers near population centres. They are also slow growing, late maturing and suffer significant mortality from barotrauma when caught and released from deep water. These factors combined mean that any additional impacts from other factors such as climate change could result in significant population declines.

Critical data gaps and level of uncertainty
Golden snapper stock structure and correlations between environmental drivers and population dynamics are unknown. Currently management allows for high harvest rates of juvenile golden snapper across most of their range. Research that better estimates the level of this catch and the effect on population viability is required.

Acknowledgements

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16.   Grey mackerel, *Scomberomorous semifasciatus*

Authors: Thor Saunders, David J. Welch and Emily Lawson

Grey mackerel, *Scomberomorus semifasciatus*, in its natural turbid water environment. Photo: David Welch.

The Fishery

> **Key points:**
> - Grey mackerel are an important commercial species across northern Australia, particularly in the Gulf of Carpentaria and western Northern Territory.
> - Commercial catches since the mid 1990s has increased dramatically in all areas except Western Australia.
> - The status of grey mackerel stocks in most Australian fisheries is considered 'uncertain'.

Also known as broad-barred mackerel, the fisheries targeting this species operate throughout tropical Australia. For a detailed description of these see Welch et al. (2009).

Western Australia

The Mackerel Fishery of Western Australia uses near-surface trolling gear from vessels in coastal areas to target Spanish mackerel (*Scomberomorus commerson*) around reefs, shoals and headlands while jig fishing is also used to capture grey mackerel, *Scomberomorus semifasciatus*. Grey mackerel have a total allowable commercial catch of 60 tonnes in each of three management areas (Kimberley, Pilbara, Gascoyne/West Coast), there are a limited number of permit holders able to access the fishery, all boats are required to have a Vessel Monitoring System and gear is limited to trolling or handlines. The northern shark fishery has also historically taken a small quantity of grey

mackerel with pelagic gillnetting. The 2011 grey mackerel catch in the Mackerel Fishery was only 13.4 t, with grey mackerel landings from the Northern Shark Fishery much lower with an annual mean catch of 2.1 t in the period from 2006/2007 to 2008/2009, noting the Northern Shark Fishery has not operated since 2008/2009 (Fletcher and Santoro, 2012). The recreational, charter and indigenous take of grey mackerel in both fisheries is unknown but is likely to be low because of the isolated nature of this coastline.

Northern Territory

The commercial Offshore Net and Line Fishery (ONLF) targets blacktip sharks (*Carcharinus tilstoni* and *C. limbatus*) along with grey mackerel. The fishery operates from the high water mark to the boundary of the Australian Fishing Zone (AFZ), although most of the effort occurs within 12 nautical miles (nm) of the coast. The fishery is managed by limited entry (17 licences permitted to operate), individual transferable effort allocations and strict gear specifications that facilitate the selective targeting of smaller, more productive sharks species, with a lesser impact on larger, less productive shark species. The fishery is managed by the Northern Territory (NT) Fisheries Joint Authority (NTFJA), in accordance with the *NT Fisheries Act 1988*.

The reported commercial grey mackerel catch increased steadily from zero in 1983 to 766 tonnes in 2003 before declining to 401 tonnes in 2010. However, from 2003-2010 there has been substantial variation in the catch of this species and operators suggest that market forces drive variations in targeting between grey mackerel and blacktip sharks (Figure 15.1). At the point of first sale in 2010 the grey mackerel component of the fishery was valued at $1.38 million (Northern Territory Government, 2011).

The estimated retained recreational catch of grey mackerel caught every year in NT has been estimated to be approximately 8,400 fish (Crofts and de Lestang, 2004; Coleman, 2004). With an assumed average grey mackerel recreational harvest weight of 3kg (usually 1-5kg) this puts annual recreational harvest of approximately 25t from NT waters (Welch *et al.* 2009). Fishing Tour Operators (FTOs) do not record grey mackerel as a species in their logsheets. However, 1446 mackerels (other than Spanish mackerel) were caught by FTOs in 2010 which equates to approximately 4t using the 3kg/fish average (Northern Territory Government, 2011). The Indigenous catch of grey mackerel is unknown but is unlikely to be substantial since this species occupies reef habitat rarely targeted by this sector.

Queensland

Gulf of Carpentaria

The Gulf of Carpentaria Inshore Fin Fish Fishery (GOCIFFF) comprises inshore (N3) and offshore (N9) net components, commercial bait netting (N11) and recreational, Indigenous and charter boat fishing within the Queensland jurisdiction of the Gulf of Carpentaria (DEEDI 2011a). The N9 net fishery harvests almost all of the grey mackerel in the GOCIFFF and operates between 7 and 25 nm offshore. Smaller numbers of grey mackerel are harvested in the N3 fishery, which mainly targets barramundi. Both net fisheries are authorised to use set mesh nets but are restricted by limited entry, allowable net length and drop and mesh size (DEEDI 2011a).

Grey mackerel are a key target species in the GOCIFFF with catches having increased consistently since logbook reporting in 1990 (Figure 15.2). In 2010 the catch of grey mackerel reached 896 t, which is the highest level of catch ever reported. The status of grey mackerel stocks in the GOC were considered to be 'uncertain' in the most recent fishery assessment and new precautionary management arrangements were proposed to be introduced in 2012 (DEEDI, 2011a).
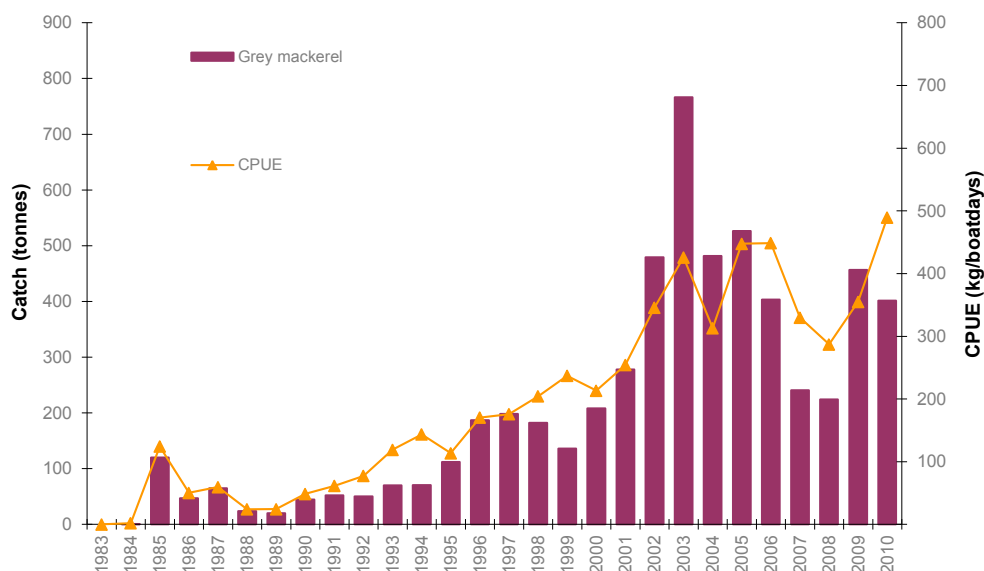
Figure 15.1. Catch and catch-per-unit-effort of grey mackerel in the NT Offshore Net and Line Fishery from 1983 to 2010.

Recreational fishers primarily use hook and line trolling methods to target grey mackerel. The most recent recreational survey conducted in the GOCIFFF during 2005 only reported on total catch so the amount of grey mackerel taken by this sector is unknown. Reported harvest of both 'grey mackerel' and 'mackerel-unspecified' by charter operators has been <1 t across the period 2004-2010 although in some years significant numbers are released (DEEDI 2011a). The Indigenous harvest of grey mackerel in the GOC is unknown but it is unlikely to be high.

East Coast
The East Coast Inshore Fin Fish Fishery (ECIFFF) comprises multi-gear commercial fisheries and recreational, charter and Indigenous fishing within all Queensland waters outside of the GOC (DEEDI 2011b). Nets represent >90% of the gear used to target grey mackerel. The number of nets permitted to be used, mesh size and length is dependent on the species being targeted and whether the fisher is operating in nearshore or offshore waters. In 2009 a total allowable commercial catch (TACC) of 250 tonnes was introduced for grey mackerel on the east coast. As a consequence the grey mackerel catch of 193t in 2009-10 was substantially lower than previous years. The stock status of grey mackerel is considered 'uncertain' due to insufficient data (DEEDI 2011b).

During the 2005 recreational fishing survey it was estimated that this sector caught 20 t of grey mackerel of which 5 t was released (DPIF 2008). Charter boats do not report grey mackerel as a

separate species in their logbooks. In 2010, this group caught approximately 15 t of unspecified mackerel (DEEDI 2011b). There is currently no information on the Indigenous grey mackerel catch in this fishery.
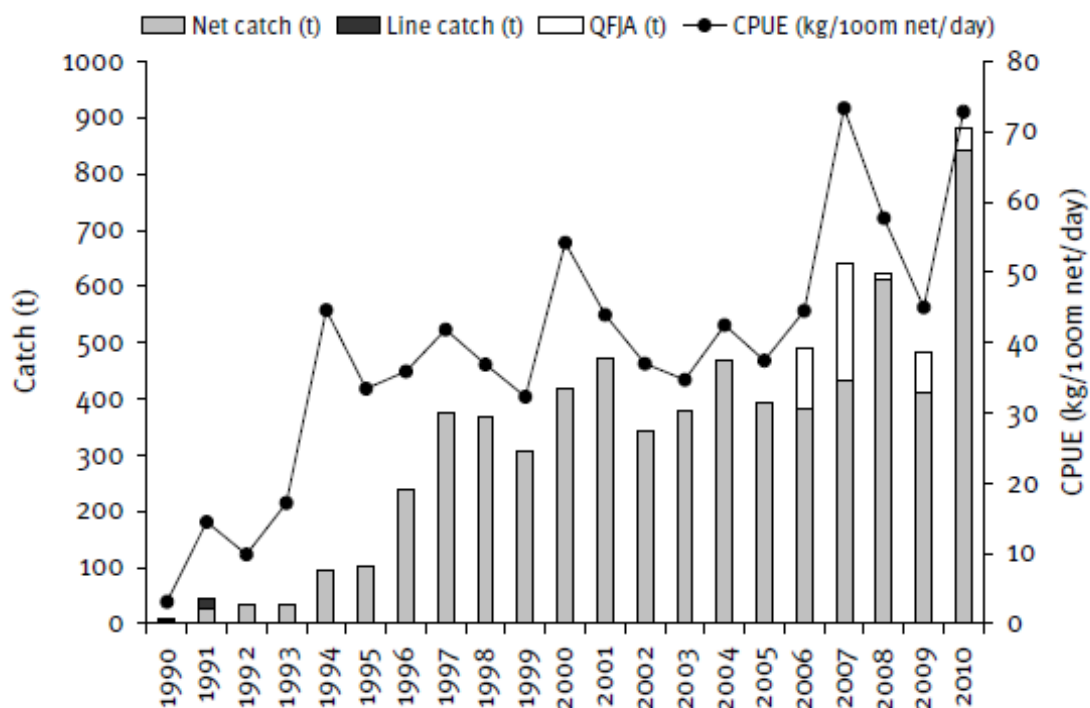

Figure 15.2. Commercial harvest (t) and catch per unit effort (CPUE) in of grey mackerel in the GOCIFFF 1990-2010 (Source: DEEDI, 2012).

Life History

> **Key points:**
> - Grey mackerel are a highly productive species with high fecundity, early maturity and quick growth.
> - They exist as at least five separate stocks across northern Australia.
> - Larval and juvenile phases are found in inshore coastal embayments and estuaries and consequently climatic factors such as rainfall and riverflow may influence recruitment.

Life cycle, age and growth

Grey mackerel, *Scomberomorus semifasciatus* (Macleay, 1884), is one of several species of mackerel (Family Scombridae). They have a rapid growth rate and can achieve a maximum weight of 10 kg and fork length of 120 cm, although the average size is between 2 and 5 kg (Crofts and de Lestang, 2004). The longevity of this species has been estimated to be up to 12 years of age, however, the majority of fish tend to be 2-4 years old. Estimates of 50 % maturity for male and female fish are 67 cm and 70 cm fork length respectively, and less than one year of age for both sexes (Welch *et al.*, 2009). Grey mackerel are highly fecund and produce more than 250,000 oocytes (eggs) per spawning (Cameron and Begg, 2002; Crofts and de Lestang, 2004). The primary spawning season runs between

August and December, however, there have been indications that some earlier spawning may be taking place in more northern regions such as north-western NT and the eastern Gulf of Carpentaria (Welch et al., 2009). Once hatched, larvae of this species move to the inner margins of coastal bays and also into estuaries (Jenkins *et al.*, 1985).

## Distribution, habitat and environmental preferences

The species is endemic to the northern Australian region and ranges from Moreton Bay in southeast Queensland, north along the Queensland coast to the southern parts of Papua New Guinea, and then west across the top of northern Australia to Shark Bay on the mid-Western Australian coastline (Charters *et al.* 2010; Collette and Russo 1984) (Figure 15.4).

Grey mackerel is a large and highly mobile schooling fish and its known preferred habitat is inshore in the often turbid waters of tropical and sub-tropical areas where they feed on pelagic baitfish consisting of sardines and herrings, and so become seasonally available to fishing operations. At certain times of the year they can also be found around rocky headlands and inshore reefs (D. Welch, pers. obs.).

While this species is found on the continental shelf it is most abundant in shallow inshore waters, often schooling around rocky reefs and underwater structures. Grey mackerel can tolerate low salinity waters and thus can inhabit nearshore areas such as river mouths and estuaries (Jenkins *et al*. 1985; Welch *et al.* 2009). Larval and juvenile life history stages of grey mackerel are found inshore, often in estuarine environments (Jenkins *et al.* 1984).

The study by Welch *et al.* (2009) identified that a number of different stocks of grey mackerel exist across the northern coast of Australia based on differences in growth, genetics, parasites and otolith stable isotopes (Figure 15.5). There is a clear separation of broad scale fishery regions between the east coast and other areas. Evidence is also provided of smaller subdivisions occurring within these areas and of minor shared stocks within the Gulf of Carpentaria (see Charters *et al.* (2010) and Newman *et al.* (2010)) (Figure 15.5).
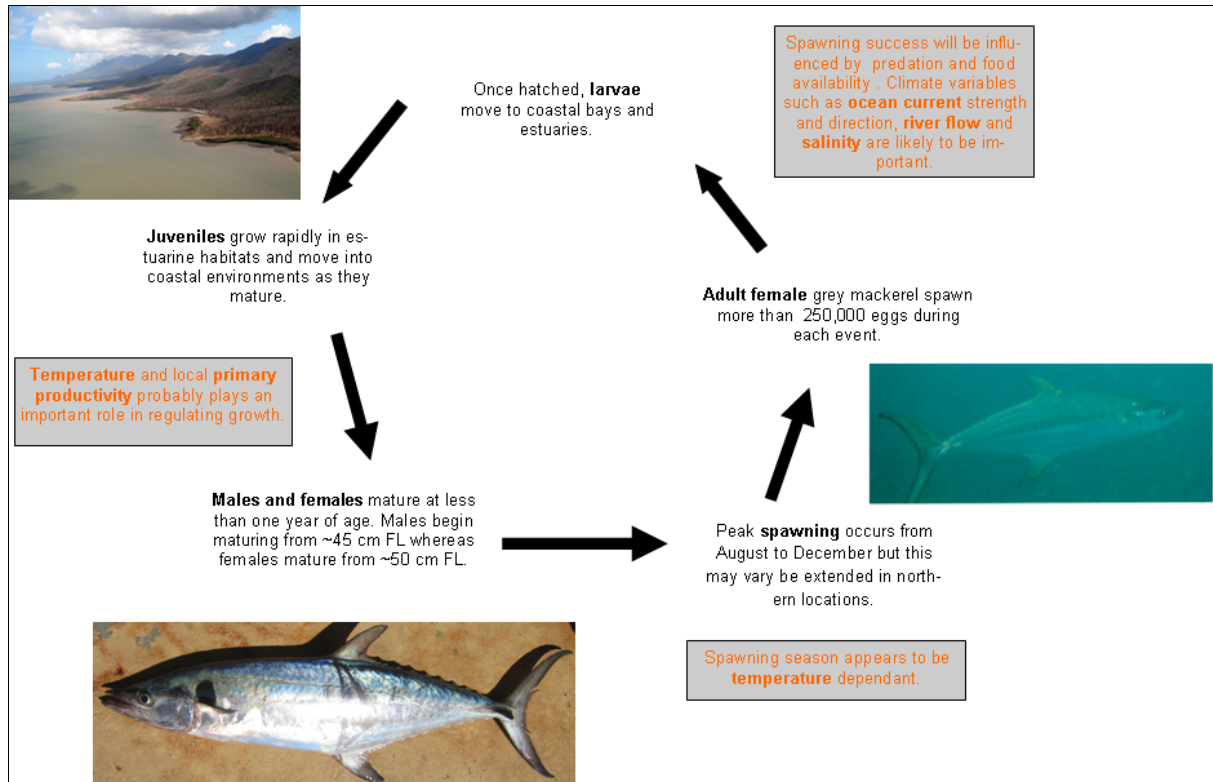
Figure 15.3. Summary of life cycle of grey mackerel and points of exposure to relevant climate change drivers. Images: D. Welch, GBRMPA.

Predators and prey

Adult grey mackerel feed primarily on pelagic baitfish such as sardines and herrings whereas larvae and juveniles feed almost exclusively on other larvae with prey sometimes reaching up to 89% of the mackerel's own body length (Jenkins *et al.* 1985; Welch *et al.* 2009).

Recruitment

Grey mackerel have a pelagic larval phase so recruitment success may be influenced by ocean current strength and direction. In addition, larvae and juveniles move into coastal embayments and estuaries so their survival is likely to be influenced by other climatic drivers such as water temperature, salinity and pH as well as rainfall and river flow.
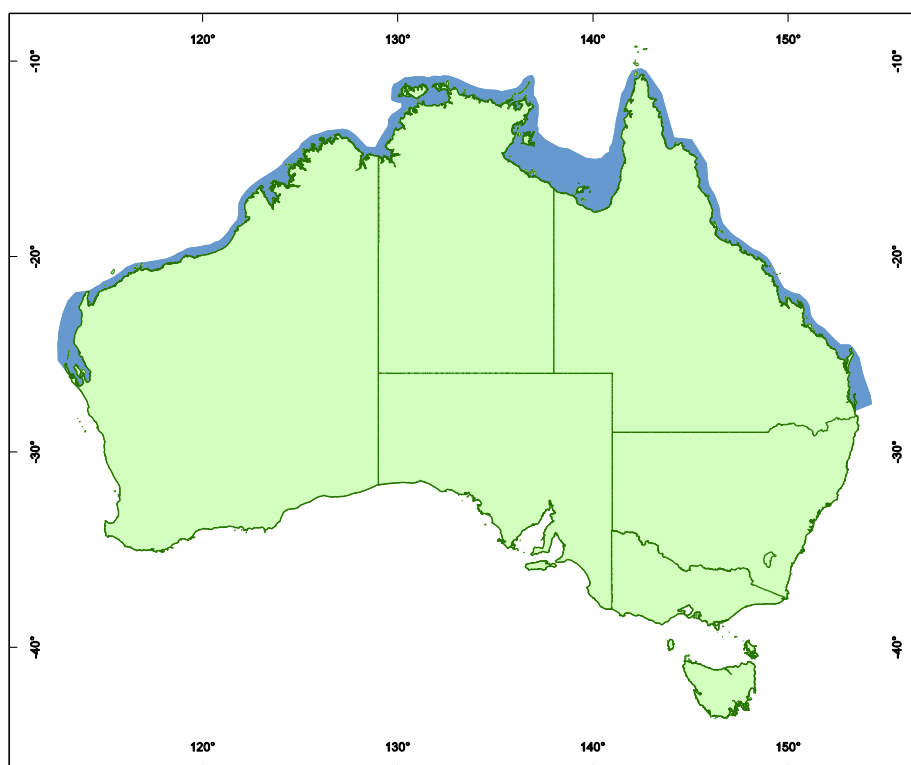
Figure 15.4. Australian distribution of grey mackerel.

Current impacts of climate change

Key points:
- Any current impacts of climate change on grey mackerel are unknown.

There are no known current climate change impacts on grey mackerel. When determining the impact of climate change on fisheries the stocks identified by Welch *et al.* (2009) should be considered separately.

Sensitivity to change

Key points:
- The sensitivity of grey mackerel to changes in environmental variables is unknown.
- Populations may be more impacted by climate change if mechanisms for stock structure inhibit large-scale migrations.
- Due to their estuarine and inshore habitats larval and juvenile phases are likely to be influenced by changes in a variety of climatic variables.

The sensitivity of grey mackerel to environmental variables is unknown. The sensitivity to climatic change by this species is likely to be related to the mechanisms driving their fine-scale stock structure. If they are structured by barriers they are unable to cross then regional changes in temperature and salinity could impact the abundance in these populations. However, if the

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mechanism that is driving the population structure does not prohibit large-scale movements then this species will be less sensitive to regional changes. Given that their larval and juvenile phases inhabit coastal embayments and estuaries these individuals may be more sensitive to changes in climatic variables such as ocean current strength and direction, water temperature, salinity, pH and rainfall and river flow

Figure 15.5. Map of northern Australia showing the approximate boundaries separating the grey mackerel stocks. Dotted lines within the Gulf of Carpentaria show where the stock division was evident and indicate the possibility of more localised stocks. Source: Welch *et al.,* 2009.

Resilience to change

Key points:
- Grey mackerel may be resilient to change in climatic variables because of their broad distribution and because they are highly productive.

Grey mackerel occur over a relatively wide latitudinal range and the species is therefore able to survive over a relatively wide temperature range. In addition, they can tolerate low salinity waters and thus can inhabit near shore areas such as river mouths and estuaries (Jenkins *et al.* 1985). They are also a highly productive species with rapid growth, early maturity and are highly fecund (Cameron and Begg, 2002; Welch et al., 2009).

Other

> **Key points:**
> - Grey mackerel are middle order predators in the food web so decreases in stocks will likely have an impact on larger predators that use them as a food source and small pelagic fish that they prey upon.
> - Substantial harvest of this species over much of its range in Australia could result in substantial declines in individual populations if climate change causes reductions in stocks.
> - The relationships between environmental variation and grey mackerel populations are unknown.

Ecosystem level interactions

Grey mackerel are second order predators in the tropical pelagic environment predating upon smaller pelagic fish species such as pilchards and herring while sharks, bill fish and Spanish mackerel would all predate upon grey mackerel.

Additional (multiple) stressors

Grey mackerel are harvested at significant levels from the east coast of Qld to the NT/WA border. Given their fine-scale stock structure any additional mortality associated with the impacts of climate change could cause significant localised depletions of populations in these areas.

Critical data gaps and level of uncertainty

It is unknown what specific impacts climatic factors have on the abundance of grey mackerel during all phases of their lifecycle.

Acknowledgements

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17.   King threadfin, *Polydactylus macrochir*

Authors: David J. Welch and Bradley R. Moore

King threadfin, *Polydactylus macrochir*. Photo; Bradley Moore.

The fishery

Key points:
- King threadfin form the second most important species of northern Australia's inshore net fisheries after barramundi.
- King threadfin may be over-exploited in Western Australia and their status will be reviewed following a formal stock assessment.
- No formal stock assessment has been conducted in the Northern Territory or Queensland.
- Evidence of over-fishing has been observed in Queensland's Gulf of Carpentaria.

Commercial Fisheries

King threadfin form the second most important species in terms of catch and value to northern Australia's inshore net fisheries after barramundi, *Lates calcarifer*. Commercial fishers typically target threadfin using monofilament gill nets. Nets are typically set from dinghies/dories in shallow tidal waters and estuaries, or staked or anchored perpendicular to the shoreline below the high water mark. Although an important fisheries species in Australia assessment of king threadfins has been hampered in the past by a lack of good information (Welch et al., 2002, 2005) which has led to an increase in research in recent years.

Historically, threadfin caught from Western Australia and the Northern Territory has been sold to local and domestic markets as frozen fillets. Recently, interstate markets have become aware of the high quality of threadfin as a table fish and fishers are now beginning to sell whole threadfin fresh on ice to southern markets. In Queensland, the commercial threadfin catch is generally sold as frozen fillets and iced gilled and gutted fish. The majority of Queensland-caught fish are sold within the state, with smaller quantities traded on interstate markets.

In Western Australia, threadfins are targeted by commercial fishers operating in the Kimberley Gillnet and Barramundi Managed Fishery which covers a coastline from latitude 19° S all the way to the WA/NT border (Department of Fisheries, 2011). Reporting of commercial threadfin catch does not discriminate between king and blue threadfin so are collectively reported as 'threadfin'. The reported catch of threadfin from Western Australia's waters in 2010 was 83 t, which comprised 55 % of the total catch for the inshore gillnet fishery (Department of Fisheries, 2011). Most of this catch was taken from the Broome and Pilbara Coasts. Threadfin catch in Western Australia has varied between approximately 50 and 110 t from 1999 - 2010 (Figure 17.1). A minimum legal length of 45 cm TL is in effect for Western Australia. A preliminary assessment suggests that populations of *P. macrochir* in Western Australia may be over-exploited (Pember et al., 2005) though no formal stock assessment has been carried out. The data required for a formal stock assessment are currently being collected.

In the Northern Territory, king threadfin forms the bulk of the reported commercial threadfin catch, with approximately 296 tonnes harvested in 2010 (Figure 17.1) (Northern Territory Government, 2009). Changes in the distribution of commercial fishing effort have been observed over the last 15 years, with effort moving away from areas in which commercial fishing has been constrained or excluded (such and the Mary River Fish Management Zone, Kakadu National Park and the Adelaide and McArthur Rivers) to more remote areas, such as the western Gulf of Carpentaria and Arnhem Land. There is no minimum legal length requirement for *P. macrochir* in the Northern Territory. The status of populations in the Northern Territory is uncertain, with no formal stock assessments conducted in this jurisdiction.

The vast majority of the Queensland reported commercial catch of threadfins is taken by the Gulf of Carpentaria (GoC) (N3) and East Coast Inshore Net Fisheries (N1 and N2), although a small proportion is taken by hook and line across the state. The GOC N3 fishery operates from the coastline out to a distance of 7nm from the coast. Along with the Northern Territory the GoC has historically been the most important fishery region for king threadfin nationally (Figure 17.1). The bulk of catch from Queensland's GoC waters is generally taken from the south-eastern Gulf, near the population centres of Burketown and Karumba. In 2009, 289 tonnes of king threadfin were harvested from Queensland's GoC waters (DEEDI, 2009a). In 2009 there were 86 commercial fishing licences for the N3 fishery in the Gulf, of which approximately 80 were active. On Queensland's east coast, the bulk of the commercial catch is taken from around the Fitzroy River and the Narrows near the cities of Rockhampton and Gladstone. In 2009 approximately 135 tonnes of king threadfin were taken from Queensland's east coast waters (Figure 17.1) (DEEDI, 2009b).

A minimum legal length of 60 cm TL is in effect for capture of *P. macrochir* in both Queensland's Gulf and east coast waters. The status of *P. macrochir* populations in Queensland's Gulf of Carpentaria and east coast waters is uncertain, with no formal stock assessments conducted in these jurisdictions. However, evidence of overfishing has been observed in Queensland's Gulf waters, with significant age truncation and reductions in length and age at sex change compared with samples collected 10–15 years ago (Moore, 2012).
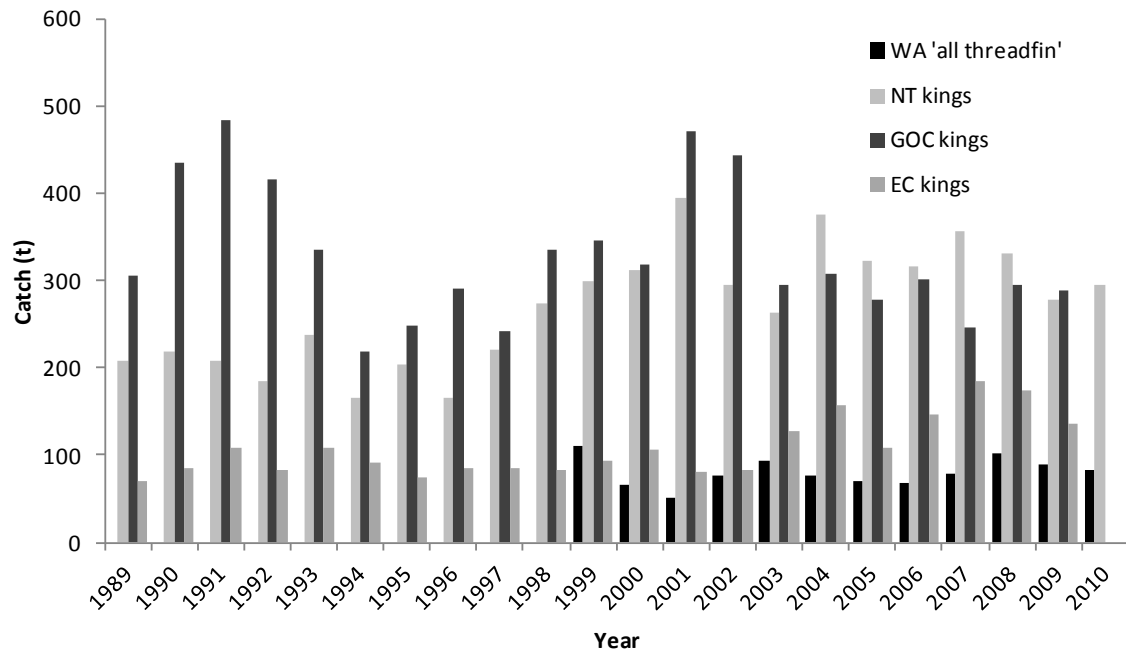
Figure 17.1. Commercial catch of king threadfin from for Western Australia (WA) which includes blue threadfin (1999 – 2010), Northern Territory (NT; 1989 – 2010), Queensland Gulf of Carpentaria (GOC; 1989 – 2009), and the Queensland east coast (EC; 1989 – 2009). Sources: Department of Fisheries, 2011; Northern Territory Government, 2011; DEEDI, 2011a, 2011b). (NB. From 2006 catch figures for the Queensland EC are reported by financial years, ie. 2006/07).

Recreational Fisheries

Recreational anglers catch king threadfin throughout the species' distribution, although fishing pressure is greatest on accessible coastlines and estuaries near population centres. Although historical information on catch and effort for the recreational fisheries across the various Australian states is limited, it is accepted that recreational fishing for threadfins has increased over the years, particularly with improved access to the more remote fishing areas in Queensland and the Northern Territory. In 2000-01 it was estimated that the total recreational catch of threadfins (all species combined) across Queensland, Western Australia and the Northern Territory was 185,000 individual fish with a further 118,000 released (Henry and Lyle, 2003).

Recreational fishing regulations vary across state jurisdictions, but are typically based on spatial closures, minimum legal size limits and bag limits. In addition to the minimum legal length requirements outlined above, a recreational daily bag limit of two fish exists in Western Australia, while a bag limit of 30 fish exists in the Northern Territory. A recreational bag limit of 5 fish is in effect for *P. macrochir* in Queensland's Gulf and east coast waters. Unlike other protandrous species, such as barramundi, there is no maximum legal size for king threadfin in any jurisdiction in Australia.

Life history

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Key points:
- King threadfin across northern Australia consist of multiple stocks and exhibit highly variable demography.
- King threadfin change sex from male to female and the size and age at both maturity and sex change varies among regions.

</div>

Life cycle, age and growth

King threadfin are protandrous hermaphrodites meaning that they first mature as males and change sex during their life to become females (Pember et al., 2005). Peak spawning occurs around August to September for populations in Queensland's Gulf of Carpentaria, and between October and January for populations in Western Australia and on Queensland's east coast (Garrett, 1997; Pember et al., 2005; Moore et al., 2011). Spawning in east coast populations occurs at lower reaches of estuaries and associated coastal foreshores (Moore et al., 2011), and it is likely that the pelagic eggs require salinities near that of seawater for high survival rates (Rod Garrett pers. comm.). Little is known of the trigger for spawning, although there is some evidence to suggest it is related to water temperature and new moon phase (Pember et al., 2005).

A number of recent studies have revealed that *Polydactylus macrochir* exhibit considerable demographic variation across northern Australia, with variation in longevity, growth rates, length and age at maturity and length and age at sex change profiles over relatively small spatial scales (Pember et al., 2005; Moore et al., 2011; Moore et al., 2012). For example, 50% of *P. macrochir* at one location in Western Australia attain maturity at approximately 23 cm TL. In contrast, the length at 50% maturity for populations on the east coast of Queensland was estimated to be 85–92 cm TL (depending on the population sampled) (Moore et al, 2011), well above the current minimum legal limit of 60 cm TL in effect for these waters. In Western Australia 50% of *P. macrochir* change sex between 79 and 116 cm, depending on region (Pember, 2006), at around 4.3–6.7 years of age (Pember et al., 2005). On the east coast of Queensland, 50% of *P. macrochir* change sex at approximately 112–136 cm TL, when fish are between 7.5–9.3 years old (Moore et al., 2011).

Considerable variation in longevity has also been observed across northern Australia. In the Fitzroy River on the east coast of Queensland, *P. macrochir* is known to reach up to 160 cm TL and live for at least 22 years (Moore et al., 2011). In contrast, individuals in Western Australia rarely live for more than 10 years (Pember et al., 2005; Moore et al., 2012). In Queensland's Gulf of Carpentaria, fish over 8 years old in the commercial catch are now virtually non-existent, despite such individuals being historically recorded in this region (Kailola et al., 1993; Garrett, 1997). The observed geographic differences in demography likely reflect regional and local variation in environmental factors and fishing pressure.
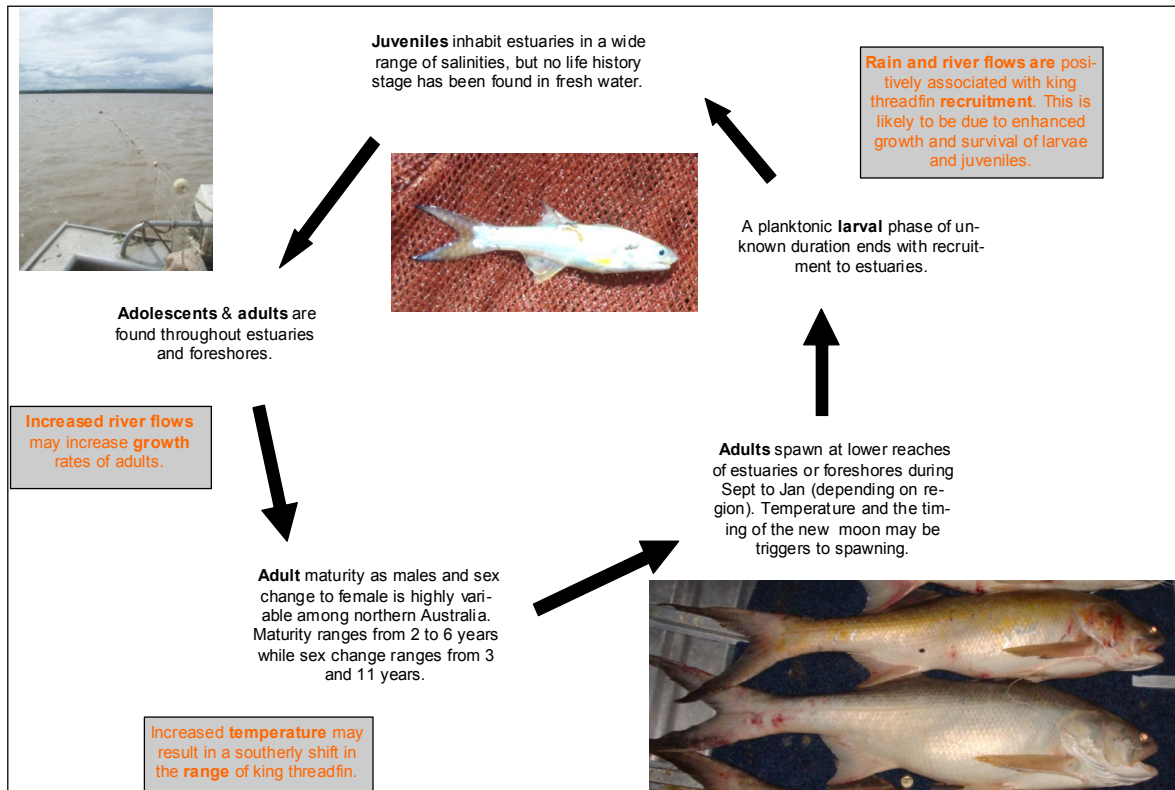
Figure 17.2. Generalised life cycle diagram for king threadfin and the stages of potential environmental driver impacts. Images: Brad Moore, Ian Halliday.

Distribution, habitat and environmental preferences

King threadfin are endemic to tropical and sub-tropical northern Australia, southern Papua New Guinea and Irian Jaya (Motomura et al., 2000; Motomura, 2004). In Australia, the species' distribution extends across tropical and sub-tropical northern Australia from the Ashburton River in Western Australia to the Brisbane region in southeast Queensland (Figure 17.3; Motomura, 2004). King threadfin inhabit estuaries and turbid coastal waters typically less than 5 m in depth (Blaber et al., 1995; Motomura et al., 2000). King threadfin do not use freshwater during any life history stage, although adults can be found upstream during winter, as saline waters intrude up the estuary (Ian Halliday pers. obs.). No king threadfin were recorded in temporary supralittoral pools in the Gulf of Carpentaria (Russell and Garrett, 1983), suggesting that king threadfin restrict their use of estuarine habitats to permanent water areas in the main channels and tributaries of creeks and rivers.

Young-of-the-year juveniles (30–100 mm FL) have been observed in north Queensland estuaries from December to May in salinities ranging from 2.0 to 37.8, suggesting a high degree of euryhalinity of these life history stages. Post-larval (i.e. juvenile and adult fish) are largely sedentary. This means that king threadfin tend to form discrete stocks over relatively small areas that are demographically, and often genetically, distinct and separate to adjacent fish (Newman et al., 2010; Welch et al., 2010; Moore et al, 2011; Horne et al., 2012). Conventional tagging data from the Australian Sport Fishing Association supports the notion of fine-scale stock structure and showed that only 4% individuals tagged in estuaries on the east coast of Queensland travelled outside of the estuaries in which they were tagged (Moore, 2012; Welch et al., 2010).
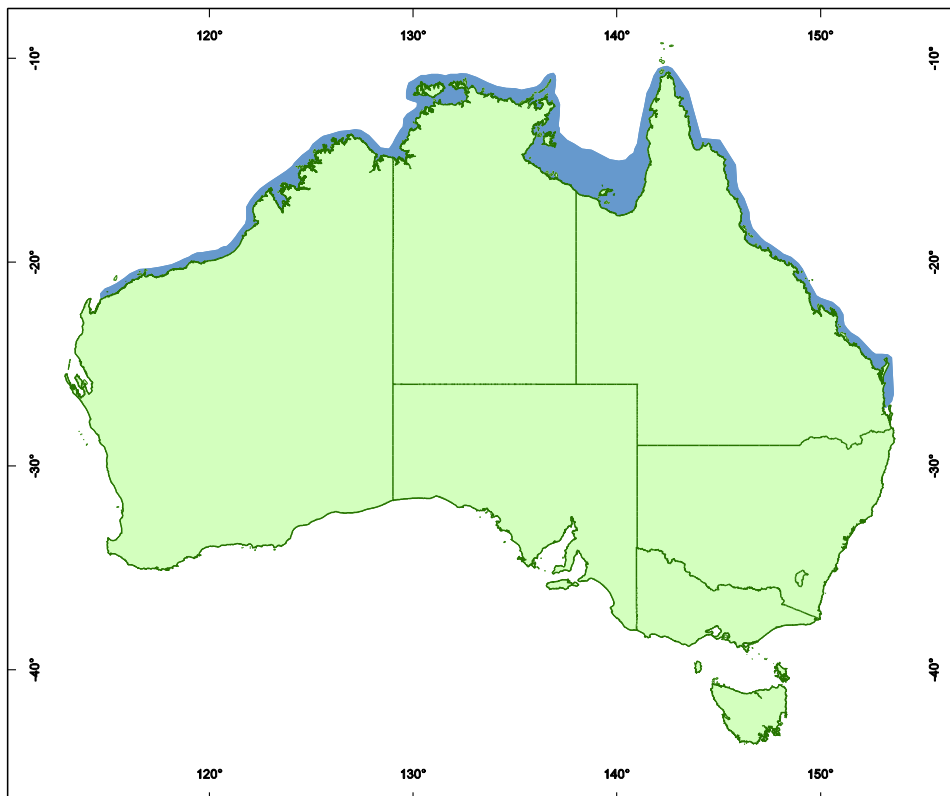
Figure 17.3. Distribution map for king threadfin.

Predators and prey

King threadfin form an important component of estuarine and coastal ecosystems, with dietary studies showing they are a significant predator of small fishes and crustaceans, in particular penaeid prawns (Brewer et al., 1995; Salini et al., 1998). Juvenile of *P. macrochir* are commonly observed in the stomachs of adult fish (B. Moore, pers. obs., Pember, 2006). Other large carnivorous fish, crocodiles and elasmobranchs prey on juvenile and adult fish (Kailola et al., 1993).

Recruitment

Little is known of the duration of the pelagic larval stage, or the sensory and swimming abilities of *P. macrochir* larvae. However, these life history stages appear to settle exclusively in estuaries (Halliday et al., 2008) or nearshore waters with estuarine characteristics (Pember, 2006), suggesting that they may be able to locate these systems, orientate themselves and take directed movements. Young-of-the-year juveniles (30–100 mm FL) have been observed in north Queensland estuaries from December to May.

Year-class strength of *P. macrochir* in Queensland estuaries is significantly and positively correlated with the timing and duration of spring and summer freshwater flow, which has been suggested to be due to greater food availability, an alteration of energy budgets in areas of decreased salinity, and/or a reduction in predation, with turbid waters enhancing juvenile survival rates (Halliday et al., 2008).

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Current impacts of climate change
There are no known current impacts of climate change on king threadfin.

Sensitivity to change

> **Key points:**
> - Reduced rainfall may depress king threadfin recruitment and growth rates.
> - Highly localised adult assemblages may be vulnerable to changes in local conditions.

King threadfin populations have been shown to be strongly influenced by rainfall and freshwater river flows (Halliday et al, 2008), although this relationship may vary regionally (Halliday et al., 2012). By examining age structures from commercial fishery catches over consecutive years they found that variation in year-class strength (as an indicator of the overall recruitment and survival of juvenile king threadfin) was consistently and positively correlated to the amount of freshwater flowing or coastal rainfall delivered into the Fitzroy River estuary in central Queensland during spring and summer. They hypothesised that this may be due to either increased biological productivity of the estuary system thereby increasing availability of food and enhancing growth, decreased salinity resulting in lowered energy budgets, or increased turbidity increasing juvenile survival through reduced predation (Halliday et al, 2008). The first hypothesis is supported by the documented evidence that major food sources of king threadfin, penaeid prawns and *Acetes*, show a significant positive correlation between catch and river flows (rainfall), however this relationship can vary regionally (Vance et al, 1985; Halliday and Robins, 2007; Meynecke and Lee, 2011).

Robins et al (2006) demonstrated that barramundi growth rates were significantly and positively correlated with freshwater flow rates (rainfall). Given the remarkable similarities between the life histories of the two species, notwithstanding the freshwater phase in barramundi (Halliday and Robins, 2007), it is very possible that king threadfin may also show increased growth rates in response to higher freshwater flows.

Resilience to change

> **Key points:**
> - Fine scale stock structure of king threadfin reduce the species resilience to localised changes that impact the stock.
> - Demonstrated plasticity in life history changes make them more resilient to changed environmental conditions.

King threadfin form discrete stocks that may be associated with river systems and therefore show fine spatial scale separation (Welch et al, 2010). This disjunct in connectivity may make individual stocks less resilient to local changes resulting in localised population effects. Conversely, king threadfin stocks have been shown to exhibit wide variation in key population traits including growth and size-/age-at-maturity and sex change (Moore et al, 2010; Moore, 2012). This demonstrates phenotypic plasticity that suggests the flexibility of populations in responding to changing environmental conditions. Although fishing pressure may affect such parameters, these characteristics may also be determined by different temperature and primary productivity regimes experienced in the respective regions of each stock (Moore, 2012).

Other

> **Key points:**
> - Climate change impacts on key prey items such as penaeid prawns will have flow-on impacts to king threadfin.
> - King threadfin are exposed to coastal perturbations and will be particularly sensitive to increased water extraction especially on the east coast where rainfall is projected to decrease.
> - Nothing is known of the thermal and pH tolerances of early life history stages.

Ecosystem level interactions

King threadfin are an important estuarine and coastal predator of small fishes and penaeid prawns (Brewer et al., 1995; Salini et al., 1998). Factors affecting productivity of lower order food web animals will affect survival and growth, and therefore productivity, of king threadfin populations.

Additional (multiple) stressors

King threadfin are closely linked with estuarine and nearshore habitats throughout their life cycle. As such they are likely to be highly exposed to and impacted by land-based influences on water quality such as agriculture, farming and development. The life history characteristics and localised stock structure of king threadfin mean they are potentially sensitive to high levels of fishing. Localised depletion of stocks from cumulative impacts are a potential risk with evidence of such a case recently documented in the south eastern region of the Gulf of Carpentaria (Moore, 2012).

Additionally, as with barramundi, where is a strong link between river flow/rainfall and population productivity, the management of water resources by authorities may influence fisheries production of king threadfin. This will be particularly pertinent under future scenarios of lower rainfall where water allocations may be preferentially directed towards human use (Halliday and Robins, 2007).

Critical data gaps and level of uncertainty

There is good evidence that king threadfin populations are influenced by river flows and rainfall (Halliday et al., 2008). However, given the localisation of populations and the high level of uncertainty in downscaled climate predictions, future climate impacts on king threadfin are highly uncertain, especially the effects of increasing temperature and decreasing pH on early life history stages. Further uncertainty will be due to the effects of cumulative impacts and food web interactions.

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18.   Mangrove jack, *Lutjanus argentimaculatus*

Authors: Richard J. Saunders and David J. Welch

The mangrove jack, *Lutjanus argentimaculatus*, is a member of the family Lutjanidae (the tropical snappers). The species has a wide distribution in the Indo-West Pacific from East Africa, the Red Sea and east to Samoa. It has also invaded the eastern Mediterranean via the Suez Canal. The species occurs throughout the northern half of Australia from the northern half of Western Australia throughout the Northern Territory and Queensland into central New South Wales, and sometimes as far south as Sydney. Juveniles and sub-adults are found in nearshore reefs and islands, coastal estuaries and freshwater streams. Adults tend to migrate further offshore to reefs and occur to depths of at least 180 m. The mangrove jack is a particularly significant species for recreational fishers throughout its Australian distribution particularly in nearshore environments.

The fisheries

- Mangrove jack are not a major target species for commercial fisheries in Australia but are captured as by-product species in reef line and trap fisheries and barramundi net fisheries.
- They are a significant target species for recreational fisheries throughout their northern Australian range particularly in riverine and coastal areas.

Western Australia
Mangrove jack is not a major component of any commercial fisheries in Western Australia with a total of 8 t landed across the state's commercial fisheries in 2010 (Department of Fisheries, 2011). There are no estimates of recreational harvest for Western Australia. Traps are used to capture this species off the northern coast.

Northern Territory

Mangrove jack is not a significant part of any commercial fisheries in the Northern Territory. They are, however, recognised as a by-product species of recreational fishers targeting barramundi. A small number are taken in the Aquarium Fishing / Display fishery: 281 individuals in 2010 (Northern Territory Government, 2011). There are no estimates of the recreational harvest for the Northern Territory.

Queensland

Mangrove jack is captured as by-product species in the Queensland Coral Reef Fin Fish Fishery. No data on numbers or catch weight is published for this fishery however it is likely to be insignificant (DEEDI, 2011a). The species is also landed as part of the East Coast Inshore Fin Fish Fishery (ECIFF) which has both net and line sectors but is < 1% of the total catch by weight (Simpfendorfer et al., 2007). Catch in the ECIFF has been 2, 7, 12 and 5 t for the 2006/07 – 2009/10 financial years respectively (DEEDI, 2011b). Estimates of recreational harvest by number in Queensland are 117,000, 107,000 and 77,000 for the years 1999, 2002 and 2005 respectively with similar numbers recorded as released (McInnes, 2008). There may be an underreporting of the total harvest in the commercial logbook scheme as this species is often reported in generic categories such as mixed reef fish. This species is also caught incidentally in fish trawls in the Gulf of Carpentaria.

Life history

- Mangrove jack occupies freshwater, estuaries and nearshore areas as juveniles and move offshore as adults.
- They mature late as old as 10 years or more.
- They have a very broad distribution across tropical Australia and down into temperate waters on a seasonal basis.

Life cycle, age and growth

The life history of the mangrove jack has been well investigated, particularly in Queensland (see Russell et al. 2003). This research confirmed that mangrove jack has a complex life history with juveniles and sub-adults occurring in inshore coastal and estuarine systems, and freshwater environments, with mature adults found further offshore areas (Russell et al., 2003; Russell & McDougall, 2005). Mangrove jack are a long-lived species. In freshwater and estuarine environments age estimates ranged in age from 0 to 11 years and in offshore environments from 2 to 39 years (Russell et al., 2003).

The species is gonochoristic, with mature fish primarily found in offshore environments. Males mature at a smaller size than females with a length at 50% maturity of 47 cm FL and 53 cm FL for females and can be 10 years old or more (Russell et al., 2003). Gonad development occurs between October and March with a peak in gonadosomatic index occurring in December suggesting a Spring-Summer spawning season in northern Queensland (Russell & McDougall, 2008). However, there is evidence in lower latitudes that the species spawns throughout the year (Anderson & Allen, 2001). Mangrove jack also form spawning aggregations in some parts of the world (eg. Palau: Johannes,
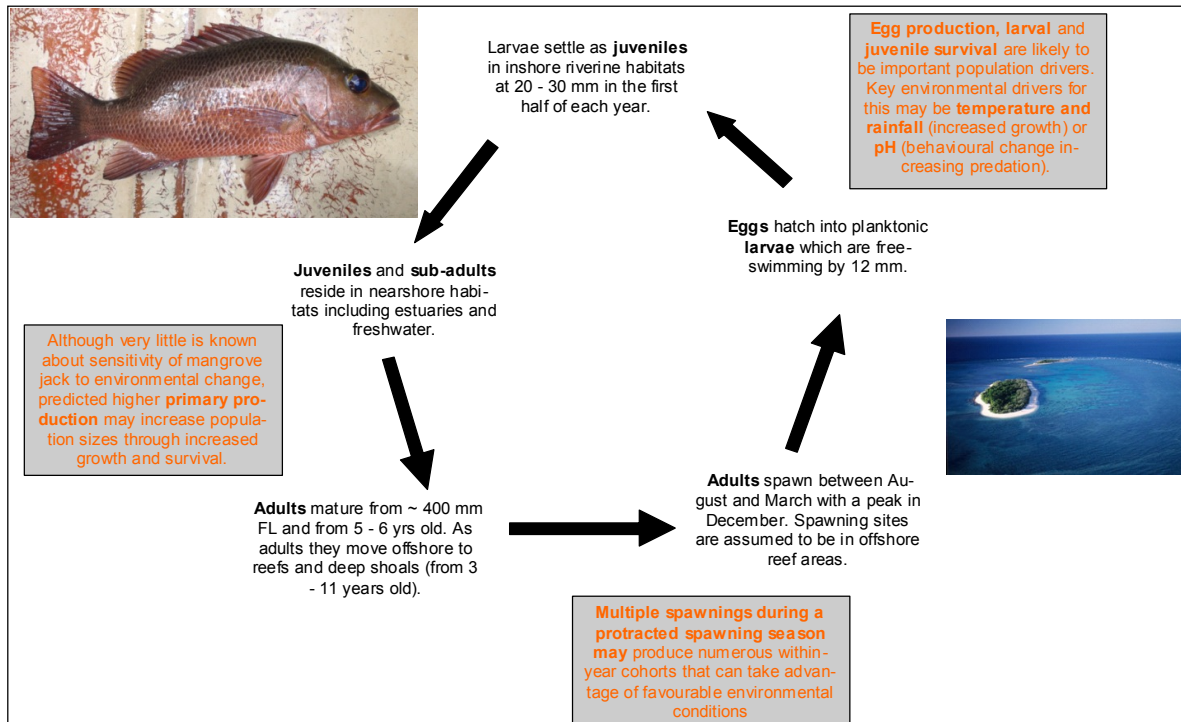
1978). In Australian waters spawning sites and behaviour are not well known although based on the distribution of mature fish it is assumed that spawning occurs offshore.

They are highly fecund broadcast spawners and larvae become free swimming by the time they reach 12 mm TL (Doi et al., 1998; Russell and McDougall, 2008). Recruitment of juveniles to inshore riverine environments occurs at 20-30 mm from February (Russell et al. 2003). Mangrove jack leave the estuarine and inshore environments between approximately 325 and 430 mm CFL at ages between three and eleven years (Russell et al. 2003).

Age and growth of mangrove jack has been extensively described by Russell et al. (2003). This study encompassed the distribution of the species within Australia but the data is best for the Queensland east coast. Some evidence for higher somatic growth rate of juveniles when able to utilise freshwater systems was identified. Furthermore, growth did vary between regions with faster growth evident in fish from northern New South Wales and southern Queensland than further north. von Bertalanffy growth parameters are provide in Table 18.1 for the Queensland East Coast.

Table 18.1. Von Bertalanffy growth parameters for Queensland east coast adapted from Russell et al. (2003). Population genetic studies across northern Australia indicate a high level of gene flow and that they are likely to belong to the same genetic stock (Ovenden & Street 2003).

Location	Sex	L_∞ (mm)	K	t_0 (years)
North of	♀	632.7	0.164	1.77
Cooktown	♂	616.2		
Ingham to	♀	673.7	0.136	1.051
Cooktown	♂	644.2		2.364
Queensland East	♀	681.2	0.126	2.893
Coast combined	♂	650.6		1.761

Figure 18.1. Generalised life cycle of the mangrove jack, *L. argentimaculatus*, and the stages of potential environmental driver impacts.

Distribution, habitat and environmental preferences

Mangrove jack occur throughout the Indo-West Pacific from Australia to southern Japan, west to East Africa and the Red Sea (Allen 1985). In Australia, it is widespread ranging from central New South Wales on the east coast to Geraldton on the west coast (Figure 18.2). It is, however, most common in the northern parts of its Australian range. The species utilises a wide range of habitats throughout its life cycle. It is commonly associated with reef environments in shallow near-shore waters to depths of at least 180 m (Kailola et al. 1993).
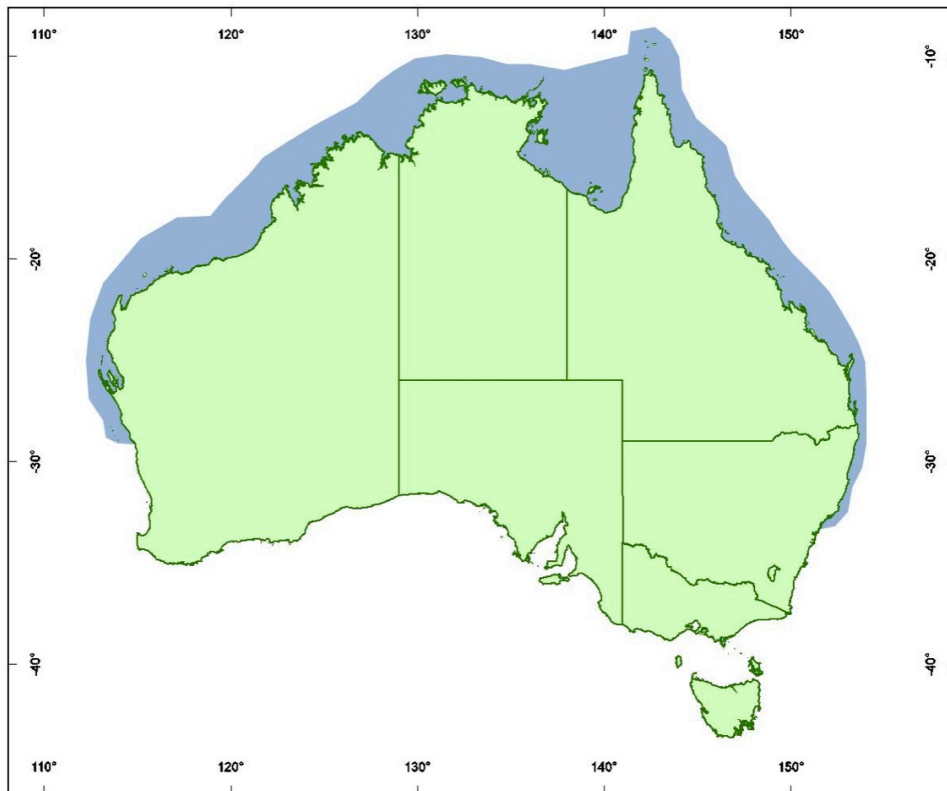
Figure 18.2. The Australian distribution of mangrove jack. This represents their usual occurrence and sometimes they can be found outside this range.

Predators and prey

Mangrove jack are carnivorous. As juveniles in creeks, mangrove jack take fish (Robertson & Duke, 1990) but crabs, particularly *Sesarma* sp., are also a significant component of the juvenile and sub-adult diet (Sheaves & Molony, 2000). As adults' mangrove jack diet is poorly documented however as a large reef predator is likely to comprise largely of a variety of fish species.

Recruitment

Young of the year mangrove jack recruit seasonally to estuaries and rivers in the first half of each year (Russell et al. 2003). Significant inter-annual recruitment variation occurs and this can occur at a large spatial scale. For example, recruitment in 2000 was generally poor for mangrove jack in several large river systems in far north Queensland across a large geographical range (Russell et al. 2000). The reasons for such inter annual variations are unknown.

Current impacts of climate change

Currently, there are no documented impacts of climate change available. A study in the US on the gray snapper, *Lutjanus griseus*, found that population sizes had increased over a 30 year period and was correlated with increasing water temperatures in estuaries. This was suspected to be because of increasingly higher winter water temperature minimums due to changes in the North Atlantic Oscillation. They postulated that the lower winter temperatures provide favourable over-wintering conditions for juvenile fish thereby enhancing recruitment (Tolan and Fisher, 2009).

Sensitivity to change

> - The sensitivity of mangrove jack to environmental change is unknown.

There are no documented studies on the sensitivity of mangrove jack to changes in environmental variables. One study using an ecosystem modelling approach found that over the next 50 years under plausible climate change scenarios (IPCC A2 emission scenario), primary production across northern Australia will increase. This was due to increases in nutrients and also temperature. They predicted that this would result in increases in fisheries catches by 10 % in NW Western Australia and up to 60 % in parts of the east coast region (Brown et al., 2009). The results of this study suggest that mangrove jack catches under future climate change is likely to increase, however, these predictions are not species-specific and so it is impossible to say what the future impact on mangrove jack would be.

Resilience to change

> - Mangrove jack use a remarkable array of habitats and environmental conditions during their lifetime, however they have life history characteristics that suggest their capacity to recover from population impacts is poor.

Mangrove jack occupy many different habitat types across a wide range of latitudes and therefore appear resilient to a range of environmental conditions. Furthermore, they are reported to be a single genetic stock across the entire northern Australian range (Ovenden and Street, 2003) meaning they are more resilient to localised changes.

Other

> - Mangrove jack are likely to be highly exposed to climate change impacts during their pre-adult stage since they occupy estuarine areas and associated habitats.
> - Although heavily targeted by recreational fishers, estimates of the harvest levels are unknown and are a high priority for future research.

Ecosystem level interactions

Climate change is predicted to have potentially profound effects on estuarine and coastal environments through a variety of physical, biological and ecological mechanisms (Sheaves et al., 2007). The complexities of the interaction of changes and their subsequent impacts on individual species makes sensible and accurate predictions challenging.

Additional (multiple) stressors

Mangrove jack represent a significant target species for recreational fisheries across all of northern Australia and increasing human populations are likely to increase this targeting. Mangrove jack rely on estuarine habitats for their juvenile and sub-adult life history stages and as such are likely to be impacted. Anthropogenic influences that effect estuarine environments (eg. water quality) are likely to affect mangrove jack populations however no data are available to determine the key variables of

influence nor the extent or direction of their potential impact. Gehrke et al (2011) concluded that fisheries in estuarine areas will become increasingly vulnerable to climate change, particularly temperature increases, where catchments have been modified by riparian clearing, agriculture, forestry or mining.

Critical data gaps and level of uncertainty
Estimates of recreational harvest are very poorly known for mangrove jack despite being a major recreational fisheries target species. Better estimation should be a key future research priority. The sensitivity of mangrove jack to environmental influences, particularly those relevant to estuarine habitats, should be investigated. Key variables of interest include temperature, rainfall, sea level rise, acidification and extreme events (Sheaves et al., 2007).

Acknowledgements

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19. Red Emperor, *Lutjanus sebae*

Authors: David J. Welch and Richard J. Saunders

An adult red emeperor from the Great Barrier Reef. Photo: Mick de Rooy.

The red emperor, *Lutjanus sebae*, is a member of the family Lutjanidae. The species has a wide distribution from east Africa to the western Pacific. It occurs in the northern half of Australia from mid Western Australia, across the Northern Territory and down the Queensland coast, primarily in reef habitats. The red emperor is a significant species for both recreational and commercial fishers throughout its Australian distribution.

The fisheries

- Commercial catches are taken in Western Australia, Northern Territory and Queensland.
- The majority of the Australian catch is by commercial and recreational fishers in the Queensland Coral Reef Fin Fish Fishery and in north-western Western Australia.
- Recreational catch levels of red emperor are poorly understood in all northern Australian fishery jurisdictions.

Western Australia

Red emperor is an important commercial and recreational species in Western Australia (Department of Fisheries, 2011). The species is taken as part of the Gascoyne Demersal Scale Fishery, the Pilbara Demersal Scalefish Fisheries and the North Coast Demersal Fishery with each fishery comprising recreational, commercial and charter fishing operations. In the Gascoyne Demersal Fishery red emperor are a non-target species and catches have ranged over the past 10 years from 9.8 t (2009/10) to 24.4 t (2000/01). In the North Coast Demersal Fishery catches are more significant, with red emperor being the second most important species by weight within this fishery. Total catch taken in both the Pilbara and Kimberley regions in 2010 was 308 t (Department of Fisheries, 2011).

The recreational catch was estimated for the area between Onslow and Broome in 2002 (Newman et al., 2004). In boat ramp surveys *Lutjanus sebae* ranked in the top ten fish kept in only one of the seven districts considered with an estimated 355 fish kept in the Point Samson district. By weight, however, the species was more significant with catches in top ten by weight in four of the seven districts; Dampier at 2,956 kg, Karratha at 190 kg, Point Samson at 1,309 kg and Port Hedland with 937 kg. The species did not feature in the catch of shore based fishers in the region (Newman et al., 2004). In 2010 the charter catch of red emperor in Western Australia was 12.7 t. Stocks in Western Australia are considered to be sustainably fished at current levels of effort (Department of Fisheries, 2011).

Northern Territory

Red emperor are a major part of the by-product catch in the Northern Territory Demersal fishery which mainly targets gold band snappers (*Pristipomoides* spp.), saddletail snapper (*Lutjanus malabaricus*) and crimson snapper (*L. erythropterus*) using drop lines and traps. Total commercial catch in this fishery was 208 t in 2010, down from 505 t in 2009. Red emperor contributed 3.5% (7.3 t) of the total catch in 2010. Red emperor is also taken as part of the Timor Reef Fishery which uses baited traps and vertical lines (NT Government, 2011). The only recreational estimate of red emperor catch was in 2002 by Henry and Lyle (2003) who estimated that 9.5 t were kept. Catches of red emperor in the Northern Territory recreational and Fishing Tour Operator sectors is currently considered to be negligible.

Queensland

Red emperor is captured as part of the Queensland east coast Coral Reef Fin Fish Fishery. The fishery is managed with spatial and temporal closures, size limits and gear restrictions. In 2003/04, a commercial catch quota was introduced for Coral Trout, Red Throat Emperor and "Other Species". Red emperor catch is included in the "Other Species" with the quota set at 956 t (DEEDI, 2011) and their minimum legal size limit was increased from 45 cm TL to 55 cm TL. Commercial catch of red emperor was estimated to be 104 t in the year prior to the introduction of quota (2003/04). In 2004/05 catch was 26 t and has steadily increased each year since and was estimated to be 60 t in 2009-10 (DEEDI, 2011).

The best estimates for the recreational catch in Queensland are from the DEEDI RFISH diary programs. The RFISH surveys estimate the retained catch of red emperor in 2002 was 88,000 fish and in 2005 there were 52,000 retained. If we assume an average weight of 4.45 kg for retained red emperor (as in Henry and Lyle, 2003), this equates to approximately 392 t in 2002 and 231 t in 2005. The reduction in catch over these survey periods corresponds with the timing of the increase in the MLS to 55 cm TL introduced during 2003.

The stock status is considered "uncertain" as there is limited understanding of the recreational catch and age structure of the population (DEEDI 2011).

Life history

> - Red emperor is vulnerable to over-exploitation having a low production potential; long-lived, slow growing, low natural mortality, large size and age at maturity.
> - Juveniles and sub-adults frequent inshore reefs and islands while adults prefer shoals and inter-reef areas usually > 15 m depth.
> - Red emperor has a protracted spawning season throughout their range.

Life cycle, age and growth

A series of recent publications on life history of red emperor based primarily in Western Australia have improved the understanding of the species considerably. Red emperor are gonochoristic but there is considerable growth difference between the sexes with males generally attaining a larger size than females (Newman & Dunk, 2002). Growth has been studied in both north-western Western Australia and Queensland (Table 19.1). Red emperor are capable of reaching sizes of approximately 100 cm (Allen, 1985) and can attain weights up to at least 15 kg however their growth rates are relatively slow (Table 19.1) and their asymptotic length is reached between 10 and 15 years of age, although growth can continue throughout their life (Newman and Dunk, 2002). The species is relatively long lived with the oldest reported specimen from the Great Barrier Reef being 32 years, one specimen from New Caledonia was 35 years and the oldest reported was from deep water off north-west Western Australia at 40 years (Loubens, 1980; Newman et al. 2010). The age-at-maturity for both sexes has been estimated to be approximately 8 years (Newman et al., 2001).

From a Western Australian study estimates of natural mortality for red emperor are low (0.104 – 0.122 $year^{-1}$) (Newman and Dunk, 2002). Red emperor are therefore considered to have a low production potential, being long-lived, relatively slow growing, low natural mortality, and large size and age at maturity, making them vulnerable to over-exploitation (Newman and Dunk, 2002).

Table 19.1. Von Bertalanffy growth parameters for Qld and WA Red Emperor.

Location	Sex	L_∞	K	t_o	Reference
Kimberley (WA)	♀	482.62	0.27	0.07	Newman &
	♂	627.79	0.15	-0.60	Dunk 2002
Queensland	♀ & ♂ combined	792.1	0.14	-0.92	Newman et al. 2000

Multiple stocks of red emperor have been found to occur along the west coast (Stephenson et al. 2001) and it is likely that multiple stocks are present across northern Australia. However, a lack of genetic difference within or between the east and west coast of Australia suggests the widespread dispersal of red emperor larvae resulting in high levels of gene flow (van Herwerden et al 2009), since adults exhibit little movement (Stephenson et al. 2001).

On the GBR red emperor have an extended spawning season of approximately 7 months duration during the Austral spring-summer period (McPherson et al, 1992) while in the Northern Territory

females are reported to spawn year round with males only spawning at limited times (Kailola et al, 1993). They are known to be broadcast spawners with a pelagic larval phase (Allen, 1985).

## Distribution, habitat and environmental preferences

Red emperor occur throughout the Indo-West Pacific from Australia to southern Japan, west to East Africa and the Red Sea (Allen 2009). In Australia it occurs from northern NSW around the northern Coast to as far south as Cape Naturaliste in south-west WA (Figure 19.2) (Newman et al. 2010). It is, however, most common in the northern parts of its Australian range. It is most commonly associated with reef environments in shallow near-shore waters to depths of at least 180 m (Kailola et al. 1993). On the GBR juveniles and sub-adults were frequently observed in nearshore habitats. Cross-shelf differences were also observed in their relative abundance with significantly more red emperor present on inshore reefs, mid-shelf reefs, and inter-reefal shoals compared with outer-shelf reefs. They were also more likely to be found in depths greater than 15 m (Newman and Williams, 1996). They are commonly associated with habitats that have both sandy and hard substrate types.

Predators and prey

A study in the Gulf of Carpentaria during 1990 found that the most common prey item by weight found in stomach content samples of red emperor were teleosts (73.0 %). The next most common by weight was crustaceans (14.1 %; not including Penaeidae and Stomatopoda) (Salini et al, 1994). Other than the above prey types, red emperor will eat a variety of prey types with annelids, cephalopods, penaeids, stomatopods, and mollusc found in stomach content samples. However, the size of the red emperor sampled during this study (n = 113) did not exceed 387 mm SL and it is possible that diet will change as fish get larger. Predators of red emperor are likely to be those of higher order (eg. sharks) and/or much larger predators.

Recruitment

There is no published information on the recruitment dynamics of red emperor however it is likely that larval survival will be variable form year to year due to inter-annual variation in favourable environmental and biological conditions, although this may be tempered by protracted spawning seasons.
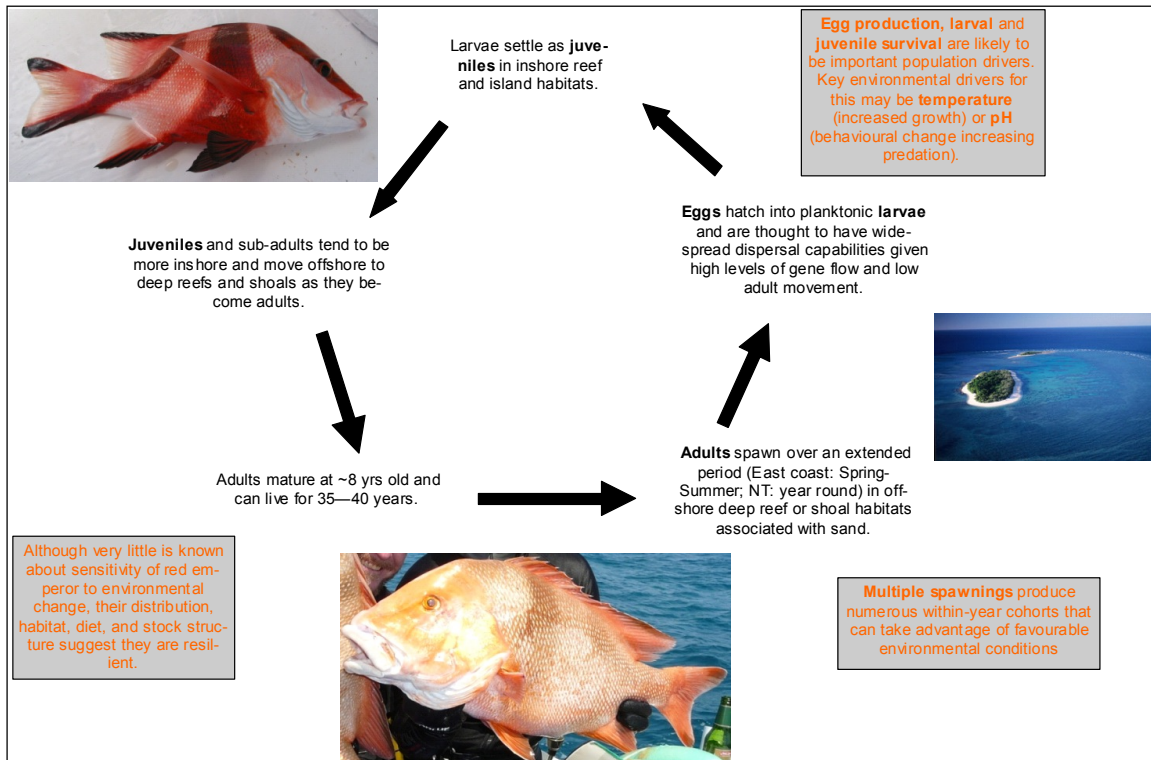
Figure 19.1. Generalised life cycle of the red emperor, *L. sebae*, and the stages of potential environmental driver impacts. Images: Michael de Rooy, GBRMPA, Fishing & Fisheries Research Centre (JCU).
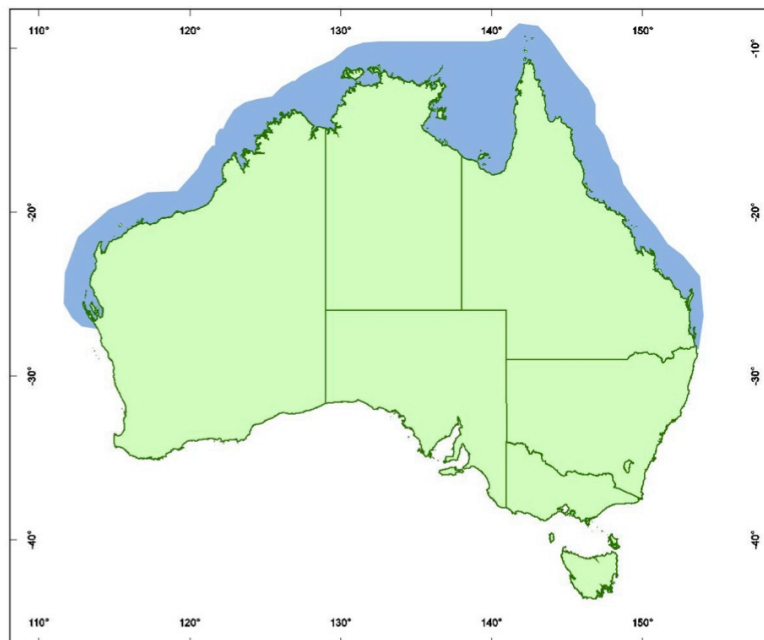
Figure 19.2. The Australian distribution of Red Emperor.

Current impacts of climate change

There are no known current impacts of climate change on red emperor.

Sensitivity to change

- Knowledge of the sensitivity of red emperor to environmental variability is very poor.

Juveniles appear to favour nearshore environments (though not exclusively) and may therefore be more influenced by land-based and anthropogenic impacts. However, very little is known of the sensitivities of red emperor to environmental factors. In other tropical lutjanid species spawning seasonality has been linked to temperature and for nearshore spawners, to rainfall also (Freitas *et al.*, 2011). However, these authors acknowledged that there was latitudinal and species-specific variation in apparent spawning patterns making generalisations about environmental spawning cues for lutjanids difficult to make. The cues for red emperor may be varied and have wide ranges since they have a protracted spawning season.

Resilience to change

- Red emperor has a broad distribution across many different habiata types and environmental conditions. They also have a protracted spawning season. These characters suggest a high potential resilienc to climate change.
- Low productivity potential represents a low resilience character however.

Red emperor is found over a wide latitudinal and temperature range and their distribution extends across the continental shelf from shallow inshore waters to deep offshore waters. They appear to prefer reef/shoal habitat associated with sand but this is variable, and their diet appears to be varied. Across northern Australia they are reported to be a single genetic stock comprising multiple separate adult stocks. They are also known to be hardy in aquariums. All of these attributes suggest that red emperor are a resilient species to differences in environmental conditions and therefore change.

Other

- Key information gaps for red emperor are recreational harvest levels and the sensitivity of the different life history stages to changes in environmental variables.

Ecosystem level interactions

Juvenile red emperor are reported to be frequently found in association with sea urchins (Allen, 1985), however the significance of this is unknown.

Additional (multiple) stressors

Although current levels of fishing effort for red emperor is considered to be sustainable in WA and the NT, in Qld it is considered uncertain due to a lack of information. Recreational catch is prominent on Qld in particular and is likely to increase in the future with increasing human populations. Further, red emperor has life history characteristics that make them relatively vulnerable to over-exploitation. The discard rate for red emperor on the GBR is known to be high given the large MLS

limit, however despite often being caught from deep water post-release survival is estimated to be high (Brown et al, 2008).

Critical data gaps and level of uncertainty

Better estimates of recreational harvest of red emperor are required to better assess stock status in all jurisdictions of northern Australia. Critical gaps that need to be investigated is the sensitivity of red emperor to environmental variation including pH and temperature, particularly for early life history stages.

Acknowledgements

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# 20.	Red throat emperor, *Lethrinus miniatus*

Authors: David J. Welch and Ashley J. Williams

Red throat emperor. Photo source: GBRMPA.

Red throat emperor, *Lethrinus miniatus*, is a medium-sized coral reef fish of the Family Lethrinidae reaching a maximum size of approximately 60 cm fork length (FL) and a maximum weight of around 3 kg. They are widespread throughout the tropical and subtropical regions of the Indian and western Pacific Oceans. In Australia they are an important fishery target species on both the east and west coasts.

The fishery

Key points:
- Catches in Northern Territory and the Gulf of Carpentaria are negligible.
- Red throat emperor is taken in Western Australia but mostly in the Great Barrier Reef line fishery.
- Recreational catch is poorly estimated.
- Red throat emperor is considered sustainably fished in WA and on the GBR.

Western Australia

Red throat emperor is taken by the commercial sector in the Western Demersal Scalefish Fishery (WDSF) which uses hand lines and drop lines. The fishery is multi-species with over 70 different species taken. In 2010 there was 45 t of red throat emperor harvested which constituted ~12 % of the total catch (Department of Fisheries, 2011). The commercial fishery is limited entry and managed primarily by spatial effort restrictions and gear restrictions.

The recreational catch of red throat emperor is unknown however the 2009/10 estimate of catch of the top 15 species, which includes red throat emperor, was 155 t of which 128 t was the three fishery indicator species (West Australian dhufish, pink snapper, baldchin groper). Management is through input and output controls including spatial and temporal closures, size limits, bag and possession limits (Department of Fisheries, 2011). Estimates of catch of red throat emperor from the charter sector are not given however are less than 10 t. Assessment of the WDSF is by monitoring estimates of fishing mortality (F) for the three indicator species and is assumed to reflect the status of all other species taken in the fishery. The most recent assessment (2007/08) determined the three indicator species to be 'recovering' (Department of Fisheries, 2011).

Northern Territory
Red throat emperor is not reported from Northern Territory waters.

Queensland
For the Queensland Great Barrier Reef line fishery (RLF) coral trout are the predominant target species historically comprising approximately 50% of the total catch (Welch et al, 2008), with red throat emperor the secondary target species. The fishery is multi-species with in excess of 125 species taken, and the fishing methods used are handlines (all sectors) and rod and reel (recreational and charter), with fishers operating from small vessels on individual coral reefs usually in depths usually less than 20 m (Welch et al, 2008). The 2009-10 estimate of the commercial gross value of production of the RLF was $45 million fishery due to increased profitability with live fish, which are almost exclusively coral trout. Currently there are 369 commercial fishing endorsements for the RLF (RQ symbol) of which approximately 205 are active (DEEDI, 2011).

Prior to 2004 the commercial sector was regulated mainly by effort controls, and the recreational and charter sectors had daily or trip bag limits. For all sectors a minimum size limit (MSL) of 35 cm total length (TL) was applied to red throat emperor for all sectors. In 2003–2004 management of the fishery changed substantially with the introduction of an annual total allowable commercial catch (TACC) allocated as individual transferable quotas (ITQs) for the key fishery species groups (coral trout, red throat emperor and 'Other' species). The red throat emperor TACC introduced was 700 t. Other management changes included increasing the MSL to 38 cm TL, introduction of a seasonal (spawning) closure and new spatial closures, gear and boat restrictions, and some effort restrictions. Since the introduction of the quota management system the TACC has not been realised in any year and in 2009/10 the reported commercial harvest of red throat emperor was 267 t (38% of the TACC) (Figure 20.1) (DEEDI, 2011).

Line fishery catches of red throat emperor are known to occur from waters of the Gulf of Carpentaria. No estimates are reported however they are almost certainly negligible. Minor quantities are also taken in the Rocky Reef fishery (SE Qld) (DEEDI, 2010). Most of the Australian catch of red throat emperor is reported to come from the Reef Line fishery (RLF) on the Great Barrier Reef from commercial, recreational and charter fishing sectors.

Harvest estimates of red throat emperor by the GBR charter sector in 2009/10 was 80 t while in the recreational sector for the years 1999, 2002 and 2005 catch was estimated to be 171,000, 155,000 and 89,000 fish respectively (DEEDI, 2011). An initial stock assessment for red throat emperor was

carried out in 2006 that assessed stocks to be sustainably fished (Leigh et al., 2006). The most recent assessment also considered stocks to be sustainably fished (DEEDI, 2011).
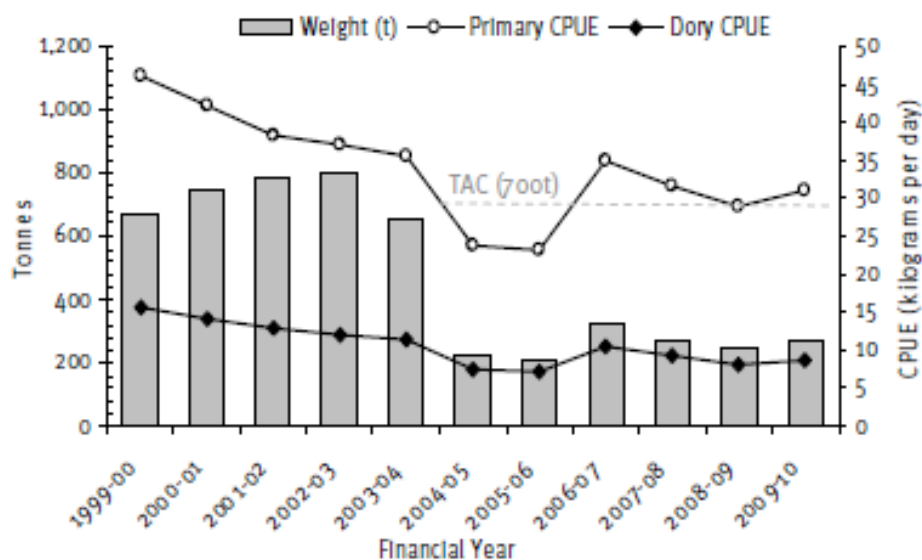
Figure 20.1. Commercial catch of red throat emperor from the Great Barrier Reef line fishery for the financial years (quota years) 1999-00 to 2009-10. Catch-per-unit-effort for primary vessels and dories are also indicated. (Source: DEEDI, 2011).

Life history

> **Key points:**
> - Red throat emperor is a moderately productive species with a narrow distribution on the east and west tropical/sub-tropical continental shelf areas.
> - Knowledge of the early life history is completely lacking, including juvenile habitat.

Life cycle, age and growth

Red throat emperor is a medium-sized coral reef fish reaching a maximum size of approximately 60 cm FL and a maximum weight of around 3 kg (Williams et al., 2003; 2007). Reports of red throat emperor reaching 90 cm FL and 9 kg in weight (Carpenter, 2001) are likely to be other emperor species, such as *Lethrinus nebulosus*, *L. laticaudis*, *L. erythacanthus* and *L. xanthochilus*, which have been misidentified.

The early life history of red throat emperor is poorly understood. The eggs and larvae have not been identified in plankton samples and juveniles (< 15 cm FL) have not been observed so their preferred habitat is unknown. Growth is better understood and is relatively fast in the first few years of life. Red throat emperor can reach their maximum size at around 6 years of age, with a maximum age in excess of 20 years (Brown and Sumpton, 1998; Williams et al., 2003; Williams et al., 2007). Patterns of growth vary significantly among regions of the GBR, with fish in the southern GBR reaching a

larger maximum size than those in the northern GBR (Brown and Sumpton, 1998, Williams et al., 2003; 2007).

Red throat emperor is a protogynous hermaphrodite, whereby individuals mature first as females before changing sex later in life (Bean et al., 2003; Sumpton and Brown, 2004). However, some of the oldest red throat emperors are female, suggesting that not all individuals change sex and highlighting the plasticity of sex change in this species (Williams et al., 2006). The peak spawning season for red throat emperor occurs between July and November on the GBR (Sumpton and Brown, 2004; Williams et al., 2006). It is not known whether there are intra-seasonal peaks in spawning associated with the lunar cycle. However, the frequency of eggs at different developmental stages in the ovaries of spawning females suggests red throat emperor are batch spawners and may spawn more than once during the spawning season (Williams et al., 2006).

The spawning behaviour of red throat emperor is not known, but occasional large commercial catches during the spawning season suggests that they may form relatively large aggregations. The proportion of females that spawn during the spawning period varies among regions of the GBR, with up to 100% of females spawning in the northern GBR and less than 43% spawning in the southern GBR (Williams et al., 2006).

Sumpton and Brown (2004) estimated that females in the Swains and Capricorn-Bunker (southern) regions of the GBR were first capable of spawning at age 3 years and 35–40 cm FL. In a more recent study, Williams et al. (2006) estimated the average size and age of mature females from the Capricorn-Bunker region to be 28 cm FL and 1–2 years. Sex change occurs over a wide size and age range, but 50% of fish become male by about 43 cm FL and 7 years of age (Sumpton and Brown, 2004; Williams et al., 2006).
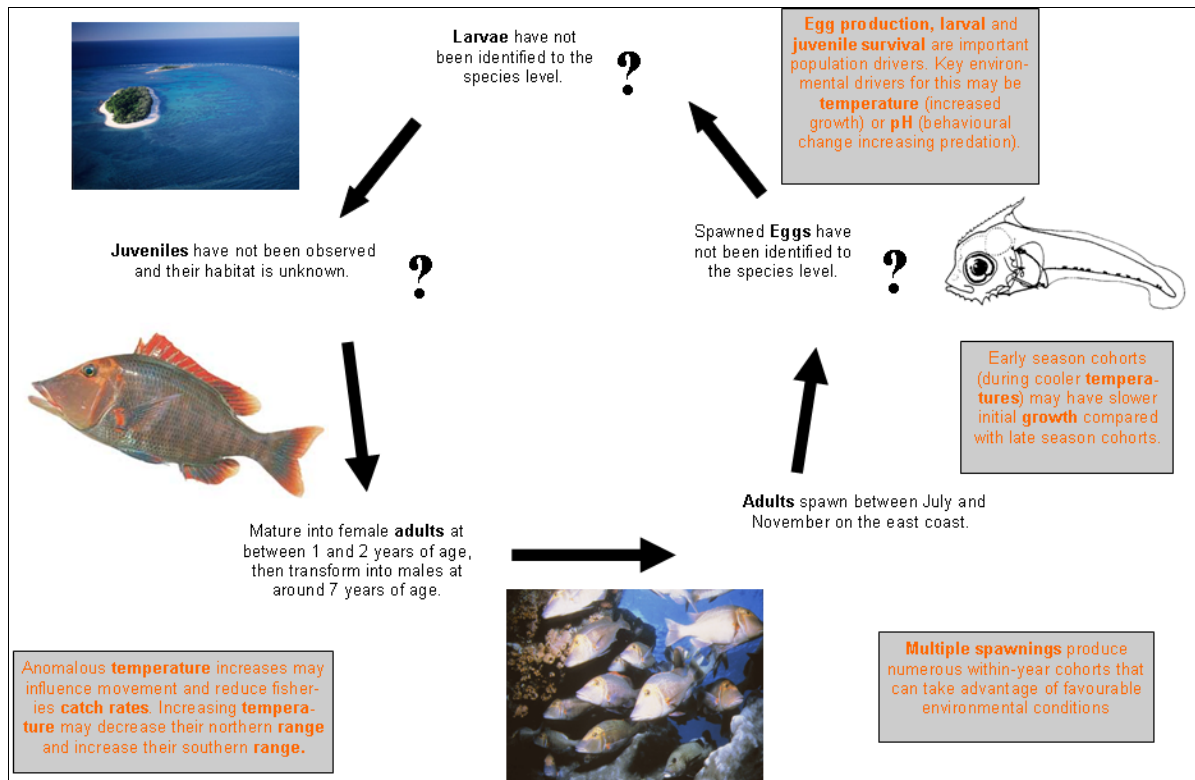
Figure 20.2. Generalised life cycle of the red throat emperor, *L. miniatus*, and the stages of potential environmental driver impacts. Images: Leis and Rennis, 1983; GBRMPA.

Distribution, habitat and environmental preferences

Reports of red throat emperor are widespread throughout the tropical and subtropical regions of the Indian and western Pacific Oceans (Figure 20.3). However, it is more likely that red throat emperor has a much more restricted distribution, as many reports of the species have been misidentifications or cannot be confirmed. Red throat emperor is confirmed to occur along the tropical and subtropical coasts of eastern and western Australia (Figure 20.3), New Caledonia, and the Ryukyu Islands of southern Japan. These confirmed reports reveal a disjunct distribution separated by the equatorial zone, and a narrow longitudinal range between approximately 110 °E and 170 °E (Carpenter, 2001).

In Australia, red throat emperor occurs along the west coast from the Dampier Archipelago in the north to the Houtman Abrolhos Islands in the south. On the east coast of Australia, red throat emperors have been found from Cooktown to Norfolk Island. However, their usual GBR distribution on the Queensland east coast lies between approximately 17 °S (Cairns) and 26 °S (Fraser Island) (Figure 20.3) (Williams et al., 2006).
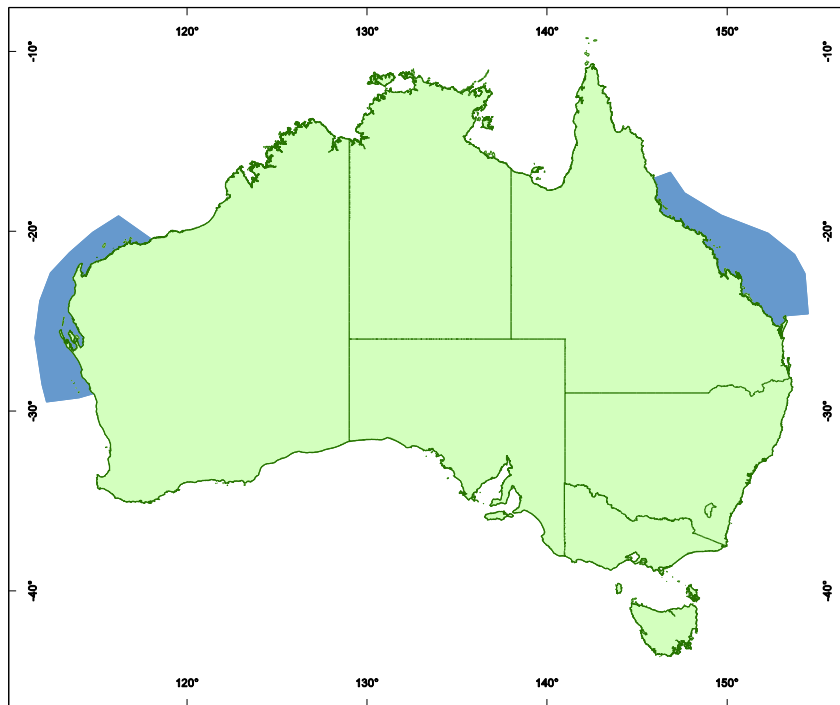
Figure 20.3. Australian distribution of red throat emperor.

Red throat emperor is a demersal species that is typically associated with coral or rocky reefs, although it is also commonly encountered on shoal and rubble habitats between reefs (Newman and Williams, 1996). Along the GBR, red throat emperor is mostly found on mid and outer shelf reefs in depths from 2 m to at least 128 m, and is rarely found on inshore reefs (Newman and Williams, 1996). There are also reports of them occurring off the continental shelf in deep water.

Predators and prey

Red throat emperor is a demersal carnivorous predator consuming mainly crustaceans, echinoderms, molluscs and fish (Walker, 1978). Within this wide range of taxa, red throat emperor appears to exercise some selective feeding, preferring particular species of crab, sand dollar and sea urchin, all of which are in relatively low abundance and are typically red or purple in colour (Walker, 1975; 1978).

Recruitment

Little is known about the early life history of red throat emperor due to difficulties in identifying emperor larvae to the species level and a lack of information about the juvenile habitat. Red throat emperor eggs are approximately 0.6 to 0.9 mm in diameter (Walker, 1975), but the appearance of larvae has not been described. The duration of the larval phase and the size at settlement is also unknown. The juvenile habitat of red throat emperor is unknown, as individuals less than 15 cm FL have not been collected or observed from anywhere throughout their distribution. Williams and Russ (1994), however, hypothesised that juveniles occur in relatively deep water (> 40 m) adjacent to coral reefs, based on the fact that juveniles have not been observed during extensive surveys of shallow reef and seagrass habitats.

Current impacts of climate change

> **Key points:**
> - The passing of cyclones has resulted in dramatic increases in catch rates of red throat emperor, apparently due to a re-distribution of populations possibly due to incursions of cool water on to reef areas.

In 2009 a very large (Category 5) tropical cyclone, Cyclone Hamish, crossed over parts of the Great Barrier Reef and was immediately followed by changes in fishery catch rates of red throat emperor in the RLF (Tobin et al., 2010). These changes varied among regions however predominantly involved increases in catch rates. In 1997 a less intense but long-lived cyclone, Cyclone Justin, also resulted in significant increases in catch rates of red throat emperor (by up to ~200%) and a northerly expansion in their usual distribution (Tobin et al., 2010). Underwater visual surveys conducted following Cyclone Hamish documented structural reef damage as high as 66 % on some reefs, however observed no change in red throat emperor abundances. For Cyclone Hamish an analysis of sea surface temperature and catch rates could not determine a clear correlation. For Cyclone Justin however, a distinct cool water anomaly was found to be the most likely driver of increased red throat emperor catch rates (Tobin et al., 2010). The impacts were spatially and temporally highly variable for each Cyclone making general statements about likely impacts of cyclones highly uncertain.

Sensitivity to change

> **Key points:**
> - Temperature appears to be an important driver of red thorat emperor populations based on the apparent effects of cyclones and their narrow latitudinal range.

Red throat emperor behaviour and movement appear to be influenced by temperature given increases in fishery catch rates and range demonstrated by Tobin et al (2010) following severe weather events and associated negative temperature anomalies, as well as their relatively narrow latitudinal range. With climate change predicting increasing water temperatures red throat emperor may show a southerly range shift in the future or be more common in deeper waters of their current range. Occasional sightings have been made as far south on the east coast as North Solitary Island near Coffs Harbour, northern New South Wales.

Resilience to change

> **Key points:**
> - Red throat emperor shows some phenotypic plasticity providing some resilience to change, however they may have narrow thermal tolerances that reduces their resilience.

Red throat emperors demonstrate phenotypic plasticity with variability in sex change documented (Williams et al., 2006). They are also concluded to be batch spawners with some evidence that females may spawn more than once during the seasonal spawning period (Williams et al., 2006).

These attributes are likely to provide red throat emperor some level of resilience in the face of climate change.

However, they do have attributes that make them less resilient to changes. They appear to have relatively restricted latitudinal distributions on the east and west coasts of Australia compared with many other key species. This is possibly determined by their thermal tolerances although they do inhabit a range of depths also. Also, despite willing to feed on a variety of prey items, they also have preferred food items that are naturally in low abundance (Walker, 1975).

Other

> **Key points:**
> - A key critical knowledge gap for red throat emperor is their early life history including larval distribution and settelement, juvenile distribution and ecology, and how climate change will affect this critical life history stage.
> - More accurate estimates of the recreational harvest of red throat emperor are also required.

Ecosystem level interactions

Red throat emperor is relatively abundant within their usual range, particularly on the GBR, and therefore adults may play an important functional role in coral reef ecosystems. The recruitment dynamics are poorly understood and, like all species with a pelagic larval phase, are likely to be influenced by spatial and temporal variability in primary productivity.

Additional (multiple) stressors

Fishing effort is a potential stressor on red throat emperor despite being assessed as "under-utilised" at current levels on the east coast, and probably at similar levels on the west coast. Current estimates of recreational harvest are poor and there is potential for increased targeting in the future particularly as human populations increase.

Critical data gaps and level of uncertainty

There are several key gaps and uncertainty in the current knowledge of red throat emperor. One of the major gaps is knowledge of their early life history, particularly larval distribution and settlement, and juvenile distribution and ecology. Very little is known on the effect that climate variables have on red throat emperor life history stages. Key variables of interest are temperature and pH and experimental studies would be beneficial for this species. Uncertain estimates of recreational harvest remains a key issue.

Acknowledgements

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21.   Scalloped hammerhead, *Sphyrna lewini*

Authors: Alastair Harry & Andrew Tobin

The fishery

Key points:
- Scalloped hammerheads are captured in a range of fisheries and at various life stages in Australian waters
- There is currently no species-specific management of scalloped hammerheads and monitoring of catch is difficult since hammerheads are rarely identified below family level.
- Complex and poorly understood behaviour, migration and stock structuring make managing this species a challenge.

The scalloped hammerhead is caught in a variety of state and commonwealth managed fisheries, where it is typically a by-catch or by-product species rather than directly targeted. Shark catch is poorly documented in many of these fisheries and hammerhead species are frequently only identified to family level. This means the occurrence of the scalloped hammerhead specifically in some fisheries is difficult to determine. No formal stock assessments have been undertaken on scalloped hammerheads in Australian waters, although there is some indication that hammerhead populations in general may be declining off the east coast of Australia (Noriega *et al.*, 2011; Simpfendorfer et al, 2011).

The scalloped hammerhead is a wide-ranging, migratory species. Since there is some level of genetic mixing with nearby countries, such as Indonesia (Ovenden *et al.*, 2009), overfishing in these areas may also affect Australia. Despite evidence for genetic mixing at large scales, evidence for fine-scale stock structuring has also been found on the east coast of Queensland suggesting a need for re-evaluation of management for this species across northern Australia (Welch *et al.*, 2010).

This species displays strong sex and size segregation, with males, females and juveniles residing in different areas and thus potentially different management jurisdictions. For example, most coastal fisheries in Australia have a bias towards catching juveniles and male scalloped hammerhead (Harry *et al.*, 2011a; Harry *et al.*, 2011b). The implications of sex-biased harvesting on this species are not well understood (Harry, 2011).

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Queensland

The largest catches of hammerheads in Queensland occur in the East Coast Inshore Fin Fish Fishery (ECIFFF) and the Gulf of Carpenteria Inshore Fin Fish Fishery (GOCIFFF)(DEEDI, 2010; 2011). The vast majority of sharks harvested in both these fisheries are in the commercial sector and are caught using gillnets. The recent introduction of a total allowable catch of 600t for sharks in the ECIFF has reduced the quantity of hammerheads caught (47t were landed in 2009/10 compared to 152t in 2008/09). Until recently the majority of the hammerhead catch in these fisheries was assumed to be scalloped hammerheads (Rose *et al.*, 2003). However, Harry *et al.* (2011b) found that great hammerhead, *S. mokarran*, was a slightly larger component of the catch by weight in the ECIFF due to its larger average size at capture. Indeed, most scalloped hammerheads caught by both these fisheries are likely to be small juveniles (<1000mm TL). For example, 11,892 hammerheads were reported in the GOCIFFF catch in 2009 for a weight of 12t an average size of < 1kg (DEEDI, 2010). Understanding trends through time are complicated by poor species-specific recording in commercial fisher logbooks. Figure 21.1 demonstrates the changes that have occurred in the Gulf of Carpentaria fishery as a result of changed logbook format and fisher education. Unfortunately, the data is not sufficiently robust to make any comment about changes through time and many fishers still group scalloped and great hammerheads in their logbook entries.
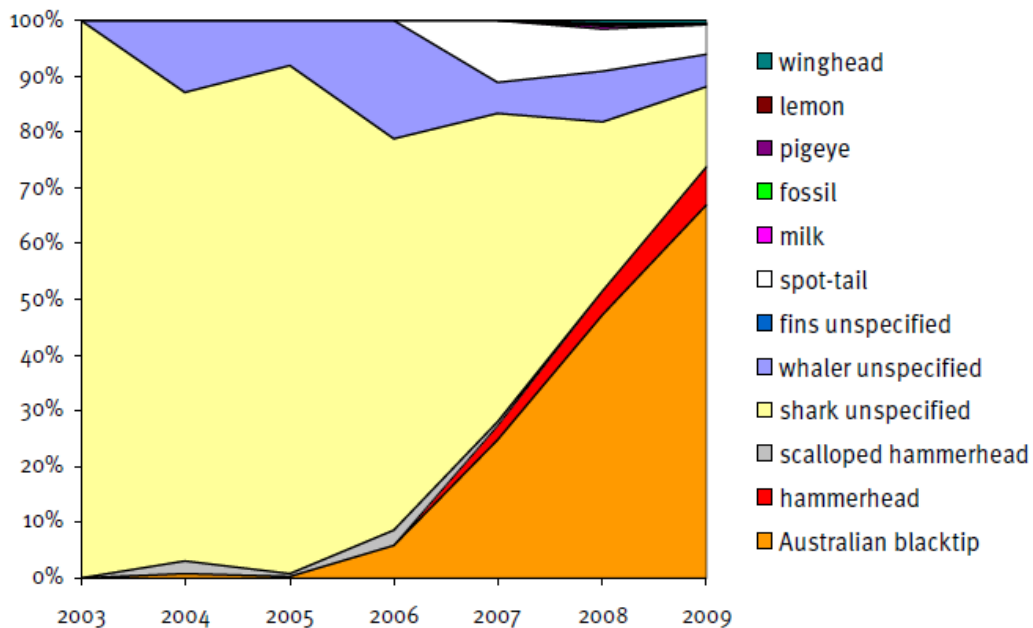
Figure 21.1. Changes in recording of shark species in commercial net fisher logbooks from 2003 to 2009 (Source: DEEDI, 2010).

Scalloped hammerhead are also caught as bycatch in Queensland trawl fisheries (Stobutzki *et al.*, 2002), however the introduction of turtle excluder devices, bycatch reduction devices and restrictions on the possession of sharks are likely to have reduced this substantially. Recreational and charter fishers also interact with hammerheads in Queensland waters (Lynch *et al.*, 2010), however size restrictions within the Great Barrier Reef World Heritage Area and current fishing

behaviour (most sharks are released alive) mean that the effect of recreational fishing on scalloped hammerheads is likely to be small compared to the commercial sector.

While the majority of Queensland's fisheries almost exclusively capture juvenile scalloped hammerheads, adults are captured and killed by the Queensland Shark Control Program (QSCP) (Noriega *et al.*, 2011). Between 2000 and 2010 the annual catch of all hammerhead species taken in the entire QSCP was between 48 and 92 individuals (Source: DEEDI, 2012). Since its inception in the 1960s, there has been a dramatic drop in the catch of hammerheads by the QSCP. For example, in north Queensland hammerheads were >50% of the catch at the beginning of the QSCP, but are now fewer than 10% of the catch (Simpfendorfer *et al.*, 2011). However, it is unclear to what extent trends in catch are related to trends in abundance, as opposed to other factors (e.g. changing gear types, localised depletion)(Simpfendorfer *et al.* 2011).

Northern Territory
The largest catch of hammerhead sharks in the Northern Territory is within the Offshore Net and Line Fishery that targets mackerel and shark (Handley, 2010). Hammerhead species made up approximately 9% (118 t) of the total catch in 2009. Although current catches are not distinguished to species level, previous observer surveys of this fishery suggests similar catch characteristics to the east coast of Queensland, with juvenile scalloped hammerheads dominating the catch, and a general bias towards catching males (Stevens and Lyle, 1989).

Western Australia
Hammerhead sharks are caught in a number of Western Australian gillnet and demersal longline fisheries termed the 'northern shark fisheries'. Due to unsustainable levels of fishing on sandbar sharks, this fishery ceased to operate and no catch has been reported since 2009/10 (Department of Fisheries, 2011).

New South Wales (NSW)
Historical commercial landings of hammerhead sharks in NSW averaged 4t between 2005 and 2010, and the highest recorded catch was 15.7t in 1993/94 (Rowling *et al.* 2010). Most of the commercial catch of hammerheads occurs in the NSW Ocean Trap and Line fishery where there has been increased targeting of sharks in the past 10 years for their fins (Macbeth *et al.*, 2009). Scalloped hammerheads make up a relatively small component of the catch in this fishery (which also catches *S. mokarran* and *S. zygaena*), and larger individuals (adult males and sub-adult females) are predominantly captured (Harry *et al.*, 2011a). Hammerheads are also captured in the NSW shark meshing program and by recreational anglers although the specific occurrence of scalloped hammerheads is not well documented (Reid *et al.*, 2011). The total number of hammerheads caught by recreational anglers and the NSW shark meshing program was around 250 sharks per year between the 1970s and 2000s (Rowling *et al.* 2010).

Commonwealth Fisheries
Commonwealth managed tuna and billfish fisheries also catch hammerheads (potentially scalloped), of which a relatively small number are retained. For example in 2006, 188 hammerheads weighing 6.2t were retained in the Eastern Tuna and Billfish Fishery (Evans, 2007), with a further 117 not kept. In 2003, 59 hammerheads weighing 833kg were retained in the Western Tuna and Billfish Fishery,

while a further 613 were not kept (Lynch, 2004). Australian tuna and billfish fisheries currently have a bycatch discard workplan and management measures for mitigating risks to sharks including a trip limit of 20 and a ban on the use of wire traces.

Illegal, unregulated and unreported (IUU) fishing
There has been a rapid rise in IUU for sharks off northern Australian waters, and this is likely to affect scalloped hammerheads (Field *et al.*, 2009; Marshall, 2011).

Life history

> **Key points:**
> - Scalloped hammerheads have a relatively low fecundity, occupy many different habitats and have a wide distribution.
> - They are a migratory species with males, females and juveniles potentially occurring in and crossing different management jurisdictions and fisheries.
> - Their general biology is poorly understood and there is considerable uncertainty in many areas of their life history.

Life cycle, age and growth
The scalloped hammerhead has a reproductive mode of placental viviparity; young are born live at 465-563 mm TL (Harry *et al.*, 2011a). Nineteen pregnant females recorded in the Queensland Shark Control Program had an average of 15 pups (range 1–28) and larger females were more fecund (Harry, 2011). Although female scalloped hammerheads appear to have an asynchronous reproductive cycle in Australian waters; pups are born year-round however there is a peak during late-spring, early summer (Stevens and Lyle, 1989; Harry *et al.*, 2011a). The timing and frequency of reproduction in scalloped hammerheads is not known with certainty. While some authors have suggested an annual reproductive cycle (White *et al.* 2008), most other large carcharhiniform sharks have a biennial or longer reproductive cycle. Pups are born in coastal estuaries and embayments and juveniles remain inshore for the first few years of life before migrating offshore. The timing of migration offshore differs between sexes (females migrate offshore earlier) and this appears to be the cause of the strong patterns in sex-segregation often observed for this species (Klimley, 1987).

There is increasing evidence to suggest that *S. lewini* is a relatively long-lived species, living to at least 30 years of age (Piercy *et al.*, 2007; Harry *et al.*, 2011a; Kotas *et al.*, 2011). However, since the majority of growth studies have used vertebrae for age determination, and none have been able to validate the vertebral banding pattern, there is a high level of uncertainty about longevity in the species. Parameters of a von Bertalanffy growth model fitted to both sexes of *S. lewini* from the east coast of Australia were L_∞ = 3,305 mm TL, K = 0.077 yr^{-1}, and t_o = -2.516 yr. Female *S. lewini* grow to at least 3,460 mm TL in Australian waters (Stevens and Lyle, 1989). Female life history is poorly documented in Australian waters; maturity in females appears to occur at lengths > 2,200 mm TL in Australian waters at an age of 10–15 years (Stevens and Lyle, 1989; Harry *et al.*, 2011a).

Distribution, habitat and environmental preferences
The scalloped hammerhead is found in tropical and warm-temperate seas worldwide (Compagno *et al.*, 2005). This coastal-pelagic species is typically found on the continental shelf from close inshore

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to well offshore, and may use deepwater and meso-pelagic habitats (to at least 1,000m) to some extent (Jorgensen *et al.*, 2009). Pups and juveniles prefer coastal estuaries and embayments for the first few years of life before migrating offshore.

In Australian waters scalloped hammerheads occur across northern Australia from Sydney on the east coast to Geographe Bay in Western Australia (Figure 21.2; Last and Stevens, 2009). Tropical coastal embayments are used as a nursery habitat for this species off eastern Australia, and neonates have been recorded as far south as Moreton Bay on the east coast. Juveniles of both sexes and sexually mature males can be found in close inshore habitats of the GBR (<25m depth) (Harry *et al.*, 2011b). The absence of females > 2 years old suggests they have begun to migrate offshore by this age. Some males do not appear to migrate offshore at all (Harry *et al.*, 2011a). Catch characteristics in the Gulf of Carpentaria and Arafura Sea are similar to the east coast of Australia (adult females absent, male-biased sex-ratio) and indicate this species may have similar behaviour across northern Australia (Stevens and Lyle, 1989).
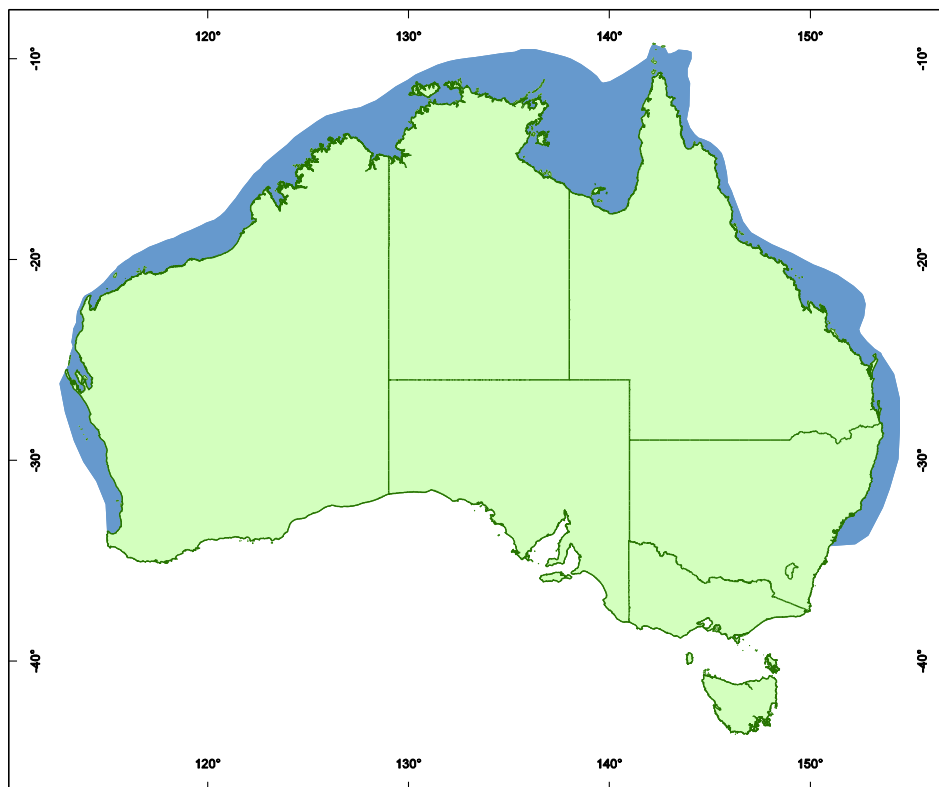
Figure 21.2. Australian distribution of scalloped hammerhead shark.

Predators and prey

This species has a broad diet that probably changes during different stages of ontogenetic development (Stevens, 1984; Klimley, 1987; Stevens and Lyle, 1989). Off northern Australia the diet of scalloped hammerheads consisted predominantly of teleost fish (80% of individuals) and molluscs (24% of individuals) (Stevens and Lyle, 1989).

Recruitment

Little is known about survival in sharks, however since young are born live and do not have a larval phase, interannual recruitment may be relatively stable and less affected by fluctuations in environmental conditions than broadcast spawners (Walker, 1998).

Current impacts of climate change

> **Key points:**
> - There are no known current impacts of climate change on scalloped hammerheads.

There has been no investigation into the effects of climate change on this species. Any changes in the distribution and abundance of the scalloped hammerhead specifically would be difficult to discern from fishery catch records, if they exist, since hammerheads are typically only identified to family level. Shark control programs in Queensland and New South Wales are the only long-term fishery-independent records of hammerhead abundance in Australia, and have not identified hammerheads to species level either (Reid *et al.*, 2011; Simpfendorfer et al, 2011).

Sensitivity to change

> **Key points:**
> - Very little is known on the sensitivity of scalloped hammerheads to environmental changes however one study suggests that higher water temperatures will effect their metabolic functioning.

Little is known about the sensitivity of shark populations to environmental changes. An integrated risk assessment for climate change on sharks in the Great Barrier Reef World Heritage area identified scalloped hammerhead as a low-risk species in both coastal habitats and shelf habitats (Chin *et al.*, 2010). The semi-quantitative assessment considered various climate change scenarios, included biological information about each species, and ranked species' rigidity and sensitivity to a variety of factors to determine their exposure to climate change. Like many large sharks, scalloped hammerhead was considered to have a relatively high adaptive capacity, and thus was not highly vulnerable to climate change. However, vulnerability increased when other synergistic factors such as fishing and coastal development were considered.

While risk assessments may give some indication of how sensitive a species is to change, there have been few experiments to provide supporting evidence. In one experiment, however, the metabolic rate of juvenile scalloped hammerheads was found to increase with temperature in Hawaii. This also increased daily food requirements and led to many animals starving during summer (Lowe, 2002). Contrary to what was expected, growth rate was determined by foraging ability and was highest when temperatures were lower. This may suggest that an increase in water temperatures could lead to greater rates of juvenile mortality in scalloped hammerheads.

Long-term fisheries datasets can also provide information on the effects of climate change on demography. While no such data has been published for scalloped hammerhead, a study of the spiny dogfish, *Squalus acanthias*, over a 60 year period found that age at 50% maturity decreased by

11 years and there was an increase in fecundity from 5.9 to 6.7 pups on average (Taylor and Gallucci, 2009). Although the authors concluded that this change was largely due to fishing rather than increasing water temperatures, the study demonstrates that many long-lived sharks may have considerable plasticity in life history traits.

Harry *et al.* (2011a) noted similar plasticity in life history traits of male scalloped hammerhead, including off the east coast of Queensland specifically. Growth rates and length and age at maturity were significantly different between two samples from two different locations off eastern Australia, suggesting that phenotype can be greatly influenced by local environmental conditions.
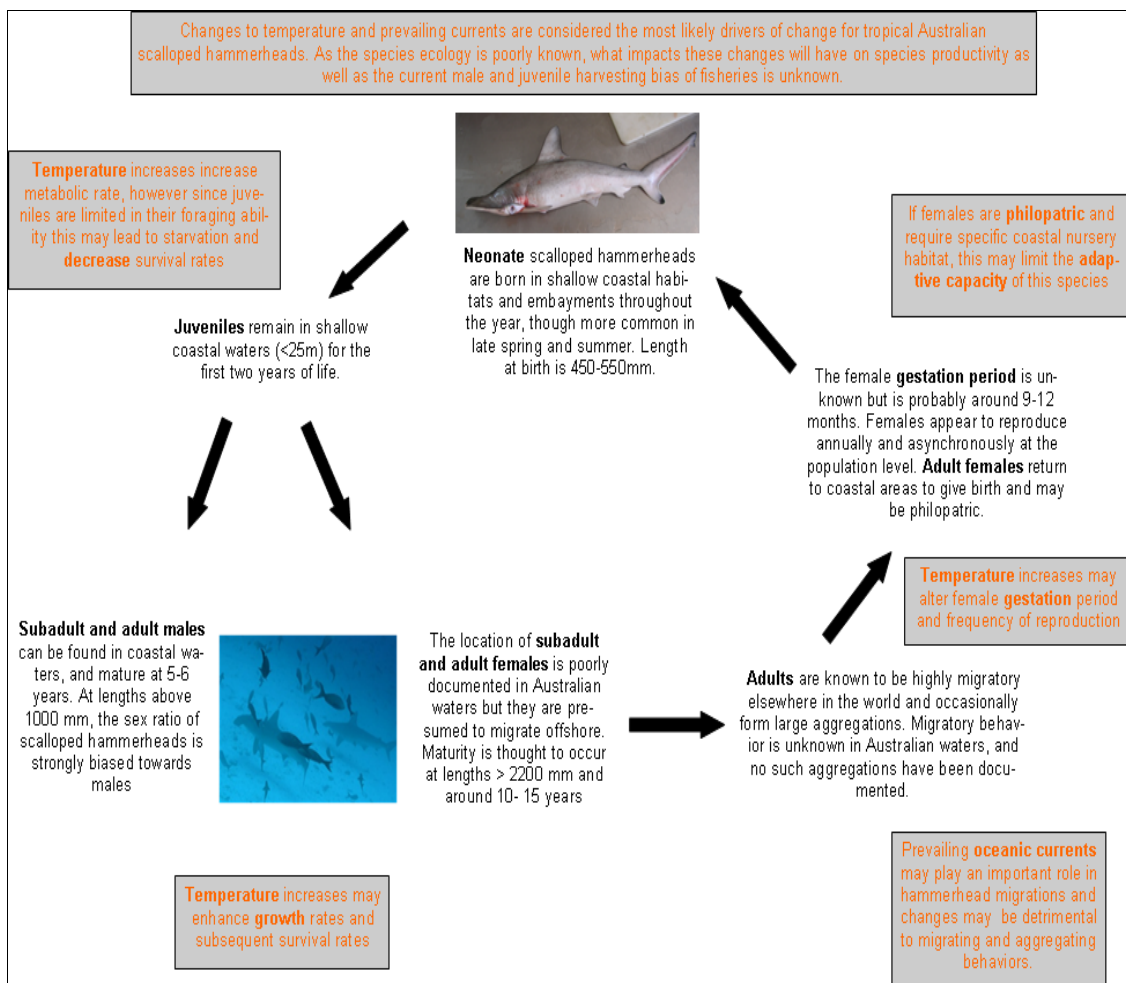

Figure 21.3. Summary of life cycle of scalloped hammerhead shark (*Sphyrna lewini*), and points of exposure to relevant climate change drivers.

Resilience to change

Key points:
- Scalloped hammerhead are likely to be resilient to climate change due to their cosmopolitan distribution across a range of habitats, as well as evidence for phenotypic plasticity.

Scalloped hammerhead sharks live in a wide range of habitats across a wide range of latitudes and through many different oceans. On this basis they are likely to be relatively robust to climate changes.

Other

Key points:
- Key information gaps exist in basic life history of scalloped hammerhead especially the validation of age and longevity.
- Fisheries monitoring and reporting programs need to begin recording hammerheads to species level to better understand background pressure on populations.

Ecosystem level interactions

Sharks constitute a major fraction of the predator biomass in tropical waters (Blaber et al. 1989, 1990a, Salini et al 1992) and as a consequence exert an important top down influence impact on tropical coastal ecosystems.

Additional (multiple) stressors

Like most large marine predatory species, sharks are vulnerable to overexploitation (Ovenden et al. 2010).

Critical data gaps and level of uncertainty

Despite its cosmopolitan distribution and abundance in many areas, the general biology and ecology of the scalloped hammerhead remains poorly understood. Since growth and longevity have never been validated in this species, it is very difficult to anticipate how populations of scalloped hammerheads are likely to respond to disturbances such as climate change and fishing. There is also a critical need for fisheries and shark control programs to begin recording catch of hammerheads to the species level.

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22.   Spanish mackerel, *Scomberomorus commerson*

Authors: David J. Welch, Thor Saunders and Emily Lawson
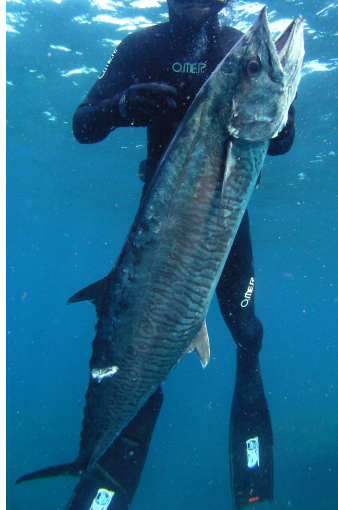
The fishery

- Spanish mackerel are a key northern Australian species with fisheries in Western Australia, Northern Territory, the Gulf of Carpentaria, Torres Strait and the east coast (Queensland & New South Wales).
- Management is generally by imposing catch limits and/or catch sharing arrangements among sectors.
- Fisheries are considered to be fully fished and/or at sustainable levels.

Fisheries for narrow-barred Spanish mackerel (referred to hereafter as Spanish mackerel) extend across northern Australia from the east coast to the west coast. Fish are taken by commercial and recreational fishing sectors by line fishing methods (see Tobin and Mapleston 2004 for a detailed description). The vast majority of the catch across Australia is taken using trolling methods using lures or baits. The fisheries are managed based on state and Commonwealth jurisdictions while also conforming to known stock structure. These key fisheries/stocks are: Queensland/New South Wales east coast, Torres Strait, Gulf of Carpentaria, north-western Northern Territory and Western Australia (Buckworth *et al* 2007). Product is sold predominantly to domestic markets.

Western Australia

In Western Australia the Spanish mackerel fishery extends from the WA/NT border and south to Perth however the majority of the catch is reported from the northwest coast. Catch from the fishery is reported separately for each of three regions: Area 1 in the north (Kimberley); Area 2 on the midwest coast (Pilbara); and Area 3 to the south (Department of Fisheries 2011). Since 2006 an

Individual Transferable Quota (ITQ) management system has been in place with a Total Allowable Commercial Catch (TACC) for each of the three Areas. The TACC for each Area is currently: Area 1 – 205 t; Area 2 – 126 t; and Area 3 – 79 t. A total of 3, 4 and 7 boats operate within each of Areas 1, 2 and 3 respectively, and boat positions are monitored by a Vessel Monitoring System (VMS). A minimum legal size limit of 90 cm total length is also applied (Department of Fisheries 2011).

The majority of the catch is from Area 1 which is a reflection of the more tropical distribution of the species. Total catch increased from 1980 and peaked in 2002 and 2003. Since then effort reductions through management intervention has caused a drop in catches to be approximately 284 t in 2010 (Figure 22.1) (Department of Fisheries 2011).
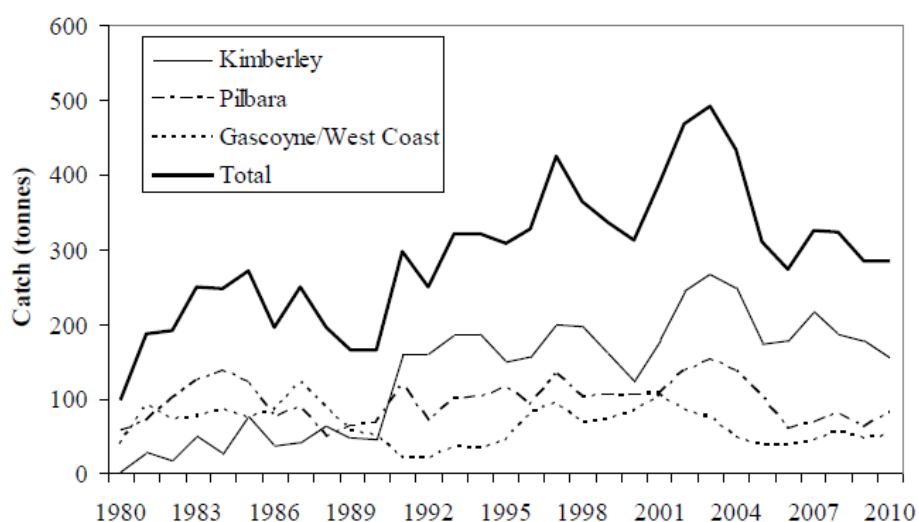
Figure 22.1. Catch of Spanish mackerel for each of the three management areas (Area 1 – Kimberley; Area 2 – Pilbara; Area 3 – Gascoyne/West Coast) for the period 1980 to 2010. Source: Department of Fisheries, 2011.
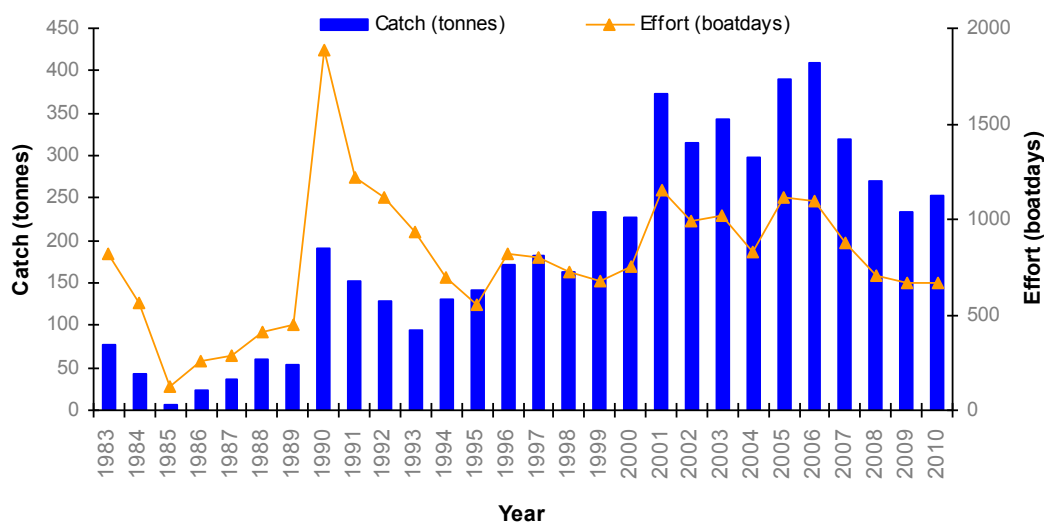
There are no recent estimates of the recreational catch of Spanish mackerel in WA however they are known to be a popular target species. Currently the effort in the fishery is considered to be at an acceptable level (Department of Fisheries 2011).

North-western Northern Territory
In the Northern Territory (NT) Spanish mackerel are taken in the Spanish mackerel fishery that operates from the coast to the outer limit of the Australian Fishing Zone and mostly around headlands, shoals and reefs. Some incidental catch is taken in the Offshore Net and Line and the Finfish Trawl fisheries (Northern Territory Government 2011). Fishers operate from a main vessel with up to two dories for each licence with each of the dories and the main vessel fishing. In 2010 there were 16 licences issued however only 12 were actively fishing. Catch is either filleted on board or trunked (head, viscera and tail removed) and stored on board. Spanish mackerel are also keenly sought-after by recreational anglers and Fishing Tour Operator (FTO) clients with most effort around the major coastal population centres of Darwin, Nhulunbuy and Borroloola (Northern Territory Government 2011). Catch limits are set for all sectors as reference points to trigger a management action if they are exceeded.

Historically, there were significant landings of Spanish mackerel by the Taiwanese gillnet fleet off northern Australia between 1974 and 1986, with annual catches perhaps as high as 1,000 tonnes in the late 1970s. Since the mid 1990s, the fishery has stabilised as a small, tightly-controlled NT-based troll fishery with a steadily increasing CPUE, possibly due to recovering populations after the period of heavy foreign harvest as well as increasing efficiency of the fishing operations (Northern Territory Government 2011).

The commercial Spanish mackerel catch has shown an increase from 1983 to 2006 with a peak of approximately 400 t and since then has stabilised at around 250 t with 254 t taken in 2010 (Figure 22.2). Catch levels have generally followed effort levels which have been influenced by price and operational factors such as availability of skilled skippers and crew (Northern Territory Government 2011).

Figure 22.2. Annual catch (tonnes) and effort (boatdays) in the NT Spanish Mackerel Fishery, 1983 – 2010. Source: Northern Territory Government 2011.

Catch from the recreational and FTO sectors are reported in numbers and recent estimates put the catch of these sectors at approximately 62 t and 15 t respectively (Northern Territory Government 2011). An unknown quantity of Spanish mackerel are released after capture by these sectors in particular and this will result in a "cryptic" mortality component of total catch levels since the release mortality rate for Spanish mackerel is estimated to be approximately 54% (Northern Territory Government 2011). Indigenous catch levels are estimated to be very low (Henry and Lyle 2003). The current low harvest compared to high catches from the Taiwanese fleet in the 1970s and 1980s suggests that current harvest rates are well within sustainability limits which is supported by none of the management trigger reference points being exceeded in 2010 (Northern Territory Government 2011).

Table 22.1. Allocation of the allowable catch among the different fishery sectors (SPM = Spanish mackerel fishery; ONL = Offshore net and line fishery; FT = Finfish trawl fishery). Source: Northern Territory Government, 2011.

Sector	Sector allocation	Weight (t)
Commercial SPM licensees	76%	342
Commercial ONL licensees	3%	13.5
Commercial FT licensees	1%	4.5
FTO licensees	3%	13.5
Recreational fishers	16%	72
Indigenous	1%	4.5
Totals	*100*	*450*

Management of the Spanish mackerel fishery in the NT is by a catch sharing arrangement based on historical catch estimates for each of the commercial, recreational, FTO and Indigenous sectors. The allowable catch level across all sectors is 450 t per year and there are trigger points for the total catch level and for catch within each sector that require management review (Northern Territory Government 2011). See Table 22.1 for the current catch sharing arrangements.

Gulf of Carpentaria

Spanish mackerel account for the vast majority of the total catch taken in the Gulf of Carpentaria (GOC) Line fishery that operates throughout Queensland GOC waters. Fishers operate generally from a main vessel with a number of smaller tender vessels (DEEDI 2012a). The fishery is managed under the Queensland Fisheries Joint Authority through the *Fisheries Act 1994* and applies a variety of input and output management controls. These include limited entry to the commercial fishery, gear restrictions, some spatial closures, and a minimum legal size limit of 75 cm TL. During 2010 there were a total of 47 licences in the fishery however only 22 were active during the reporting period. Currently the stock status is listed as 'uncertain' due to a lack of data (DEEDI 2012a).

Commercial catch of Spanish mackerel in the GOC has gradually increased during the period 2000 to 2008 where it has peaked at 287 t, while for 2009 and 2010 the catch dropped to 189 t and 183 t respectively (Figure 22.3). A significant volume of Spanish mackerel is landed as a by-product in nets set to target grey mackerel by fishers in the GOC inshore finfish fishery. In 2010, 48 t of Spanish mackerel was taken in the nets set by the inshore net fishery. During this period (2000 – 2010) catch-per-unit-effort increased (DEEDI 2012a). Most of the commercial catch in the past two years at least has come from two main fishing areas: around Weipa in the north-eastern GOC, and in the area to the northeast of Mornington Island in the southeastern part of the GOC. Effort in the charter operator sector has been decreasing over the past 5 years with catches also decreasing. In 2010 the Spanish mackerel catch by the charter sector was estimated to be 2.9 t (DEEDI 2012a).
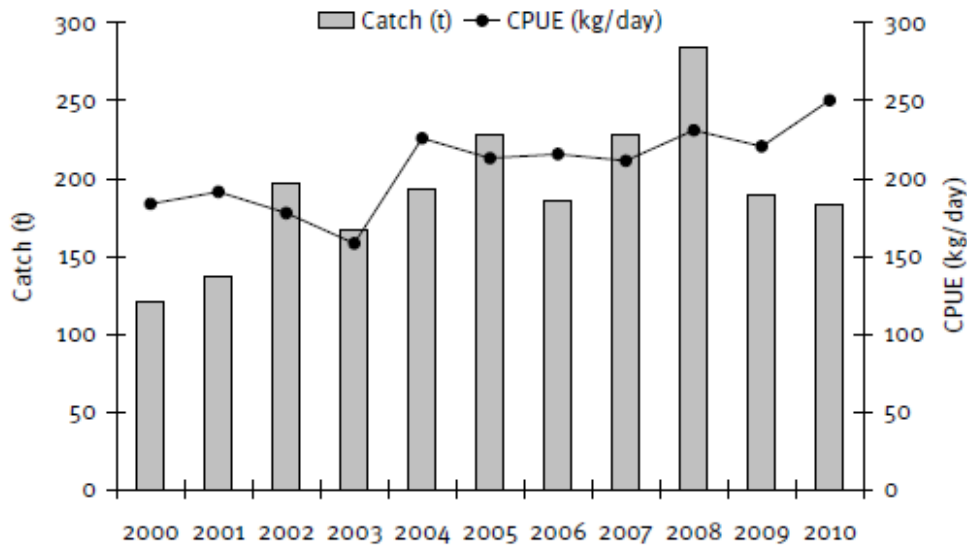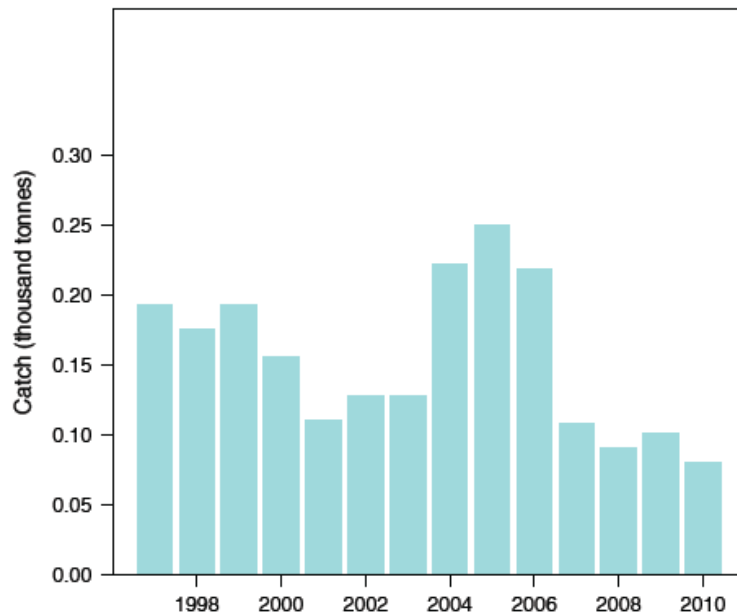
Figure 22.3. Annual total catch (tonnes) of Spanish mackerel taken in the Gulf of Carpentaria from 2000 – 2010. Source: DEEDI 2012a.

Torres Strait

Spanish mackerel is one of the key target species in the Torres Strait Finfish fishery. The fishery for Spanish mackerel is seasonal and principally targets fish near Bramble Cay in the northeastern Torres Strait between August and March (Marton *et al* 2011). The fishery is made up of fishers in two licence categories: the Traditional Inhabitant Boat (TIB) licences and the Transferable Vessel Holder (TVH) licences. As part of negotiations with the Federal Government, in 2008 all TVH endorsements were bought out and surrendered to the Torres Strait Protected Zone Joint Authority with a few still operating through leasing arrangements under quota only. In 2010 there were 5 TVH licenses with mackerel endorsements with only 4 active, and there were 161 TIB licenses with mackerel endorsements with 55 active (Marton *et al* 2011).

Despite catch restrictions, the TVH sector of the fishery accounts for most of the effort and catch and have mandatory logbook reporting requirements while the TIB sector do not. Therefore the reported catch from the fishery is always an under-estimate, as it does not include the TIB sector. Estimates of recreational catch are also not available. The most recent years of total reported commercial catch was lower than previous years reflecting the reductions in active TVH fishery licences and in 2009 was 88 t, down from its peak of approximately 250 t in 2005 (Figure 22.4; Marton *et al* 2011).

Figure 22.4. Annual total catch (thousand tonnes) of Spanish mackerel taken in the Torres Strait from 1997 – 2010 as reported by the TVH sector (non-Traditional Owners). Source: Marton *et al* 2011.

Queensland/New South Wales

On the east coast fisheries for Spanish mackerel span Queensland and northern New South Wales waters and are managed separately based on state jurisdictions despite current knowledge suggesting a single stock (Shaklee *et al* 1990). The New South Wales (NSW) fishery is highly seasonal with fish mainly only available during the summer/autumn months during a southerly migration associated with feeding and apparently linked with sea water temperatures. Commercial catches in NSW were higher during the period 1984/85 – 1993/94 (range: 15 – 51 t) and since then have tended to be lower ranging from 2 – 30 t annually (Rowling *et al* 2010). Estimates for the recreational sector are very coarse ranging from 10 – 100 t (Henry and Lyle 2003). Management in NSW uses output controls and includes a minimum size limit of 75 cm and a recreational bag limit of 5 (combined Spanish and spotted mackerel limit). NSW have adopted the Queensland assessment that Spanish mackerel is sustainably fished (Rowling *et al* 2010).

In Queensland the fishery generally extends throughout the year however historically the majority of the annual catch comes from a known spawning region near Townsville during the months of October and November (Welch *et al* 2002). The fishery is managed under a range of input and output management controls including a minimum size limit of 75 cm TL and a recreational bag limit of 3 fish. In 2004 a Total Allowable Commercial Catch (TACC) of 544 t was introduced. Since compulsory logbook reporting in 1988 reported commercial catch increased slowly to peak at approximately 800 t in 2003. With the introduction of the TACC annual reported catch has been from 200 – 400 t in the period 2004/05 to 2009/10 (Figure 22.5) (DEEDI 2012b). Reported catch from the charter boat sector has been between 20 and 30 t since 1999/00 and increased to 44 t in the 2009/10 reporting year (DEEDI 2012b).
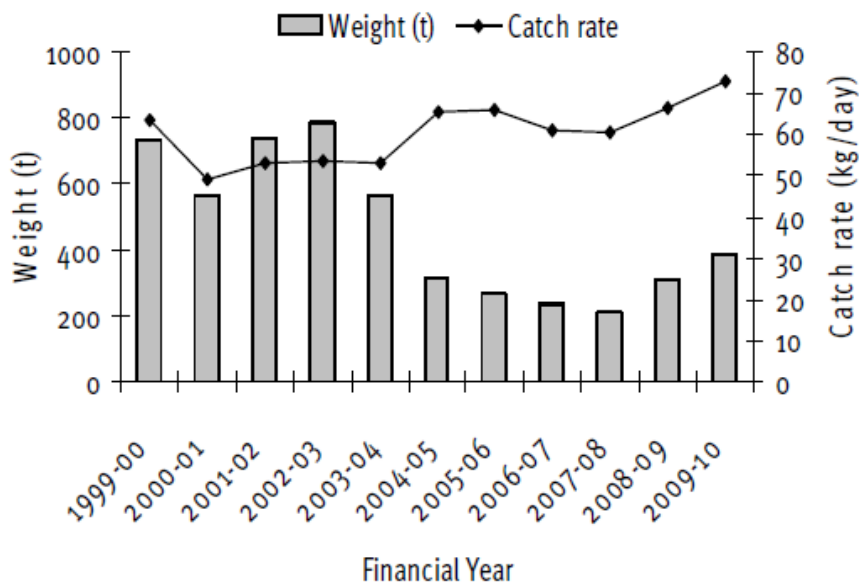
Figure 22.5. Annual total commercial catch (tonnes) and catch rate of Spanish mackerel taken on the Queensland east coast from 1999/00 to 2009/10. Source: DEEDI 2012b.

Life history

> **Key points:**
> - Spanish mackerel are highly productive being fast growing and mature at an early age. They are also highly fecund.
> - They aggregate over large areas for spawning which appears to be triggered by SST.
> - Spanish mackerel have seasonal movements possibly linked with SST, and migrate farther south during summer on the east and west coasts.

Life cycle, age and growth

Spanish mackerel are fast growing and highly mobile epi-pelagic species found in tropical and sub-tropical waters across northern Australia usually associated with reefs, shoals or current lines (Collette and Nauen 1983). Growth is rapid to a large size, with record fish exceeding 2 m in length and 100 kg in weight (McPherson 1992; Buckworth and Clarke 2001). Differential growth between sexes occurs with the females showing faster growth, greater maximum length and higher longevity reaching at least 17 years on the east coast (McPherson 1992; Tobin and Mapleston 2004) and up to 22 years on the north west coast (Mackie *et al* 2003).

Much of the following life history knowledge documented below is sourced from studies conducted on the east coast, and particularly the Great Barrier Reef. Similar life history patterns are assumed for other parts of northern Australia. Spanish mackerel are known to aggregate in large numbers to spawn. During the 1970's aggregations of spawning fish on the east coast were reported to occur between Lizard Island and Townsville. In recent years the only aggregation of spawning fish has occurred over a much smaller area on several reefs east of Ingham, north of Townsville. Fish gather

on these reefs in large numbers between September and December each year. Spawning is determined by a combination of environmental factors particularly temperature, but can be observed over much of the two month period particularly during new moon phases (McPherson 1981a). Females in pre-spawning condition are common in troll catches during the morning hours of the day of spawning. Spawning appears to take place during late afternoon and early evening during which time the fish cease feeding. Feeding behaviour resumes immediately after spawning (McPherson 1981a; McPherson 1993). *S. commerson* is a serial-spawning species and females can spawn every few nights during a spawning run. In the tropics at least, spawning may be repeated over a protracted spawning season (Buckworth and Clarke 2001). Female Spanish mackerel are highly fecund (Moltibano *et al* unpublished data).

An unknown proportion of fish older than two years of age undertake post-spawning migrations into southern Queensland and northern NSW waters on the east coast, and on the west coast a similar seasonal migration is documented. These large-scale migrations are thought to be linked to seasonal warmer currents moving southwards (Donohue *et al* 1982; McPherson 1981b). On the east coast migratory fish return northwards near to the coast and inshore islands where small localised fisheries have developed for these larger fish. Patterns in water temperature and baitfish distribution are likely to affect adult distributions throughout the year.

Eggs are released into the plankton where they develop and hatch as larvae at approximately 2.5 mm in length (Munro 1942). They develop minute teeth by the time they are 5.6 mm long and become juveniles at approximately 14.5 mm (Jones, 1962). Larval duration is two to four weeks (Ovenden 2007). Larval *S. commerson* feed almost exclusively on larval fish and invertebrates (Jenkins *et al* 1984). Larvae are found on continental shelf waters and settle as juveniles in inshore nursery grounds. The absence of Spanish mackerel larvae in coastal and estuarine habitats suggests direct movement inshore by juvenile fish rather than passive transportation of eggs and larvae by currents and tides (Jenkins *et al* 1985). Juvenile fish inhabit shallow estuaries and intertidal flats for approximately the first six months of life (McPherson 1981a). Juvenile fish between 15 and 40 cm length are found in shallow coastal waters during February and March (Williams & O'Brien 1998). By May, juveniles leave inshore areas for offshore waters where fish around 50 cm begin to be represented in the catches of commercial fishers (McPherson 1981).

Growth of juvenile Spanish mackerel is typically rapid, reaching approximately 65cm fork length (FL) in the first year. They reach the current minimum legal size early in their second year of growth and attain approximately 80 cm FL by 2 years of age. Sexual maturity for males and females occurs around 2 years of age from about 79 cm FL (McPherson 1993; Montilbano *et al* unpublished data).
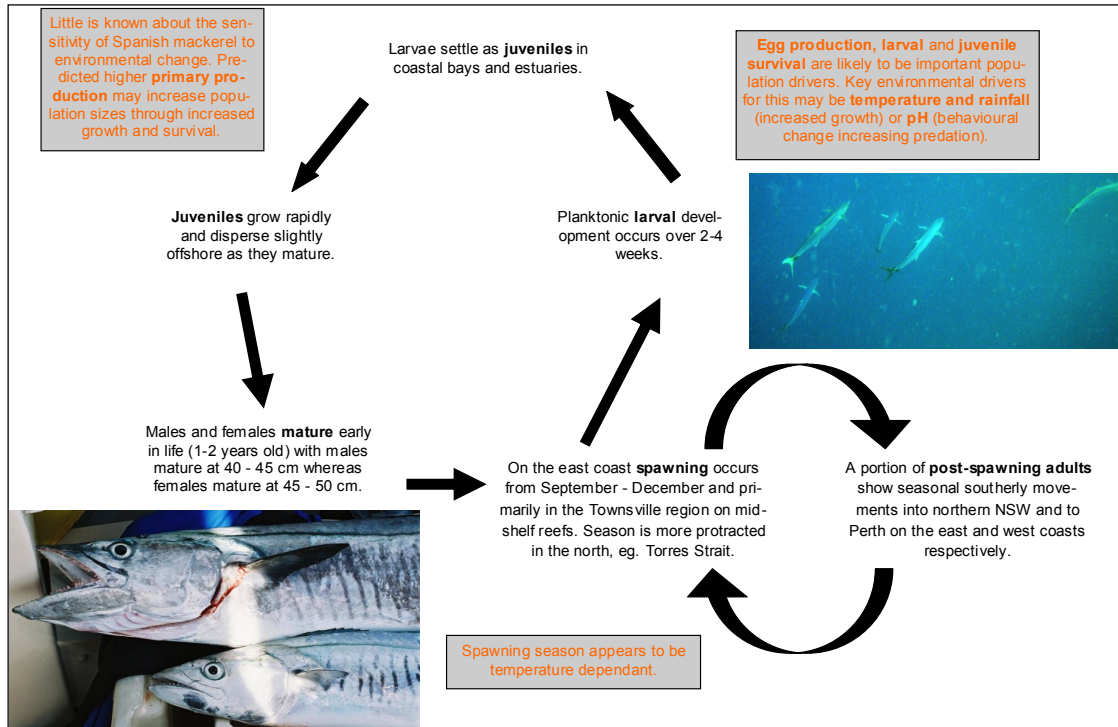
Figure 22.6. Generalised life cycle of the narrow-barred Spanish mackerel, *S. commerson*, and the stages of potential environmental driver impacts. Images: Mike Hanks, Michael de Rooy.

Distribution, habitat and environmental preferences

Spanish mackerel are found throughout tropical and sub-tropical areas of the Indo-west Pacific from Africa to Fiji. In Australia they extend across the northern coastline throughout continental shelf waters to approximately 30°S on both the east and west coasts (McPherson 1981; McPherson 1992). Their usual range has a southerly limit of the central NSW coast in the east, and at least as far south as Perth on the west coast (Figure 22.7). They are a highly mobile pelagic fish commonly associated with reef edges and headlands and have a preference for shallow coastal and continental shelf waters (Quinn 1993). They are usually associated with water temperatures of approximately 20° C or warmer (Buckworth and Clarke 2001; McPherson 1992).
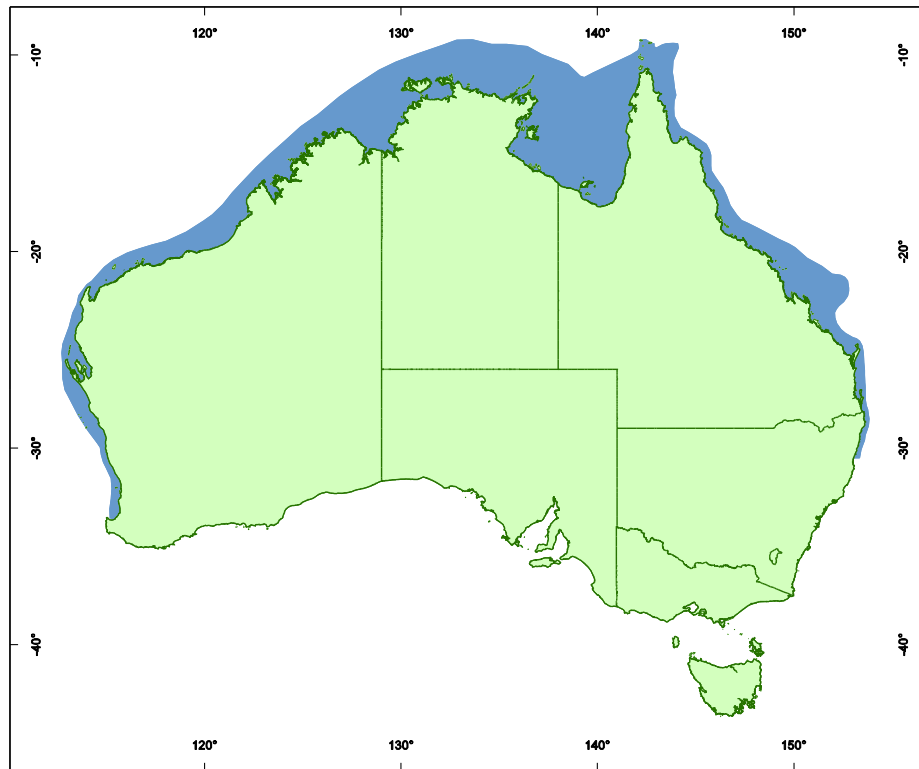
Figure 22.7. Australian distribution of Spanish mackerel.

Spanish mackerel is usually seasonally abundant (Collette and Nauen 1983). In the Northern Territory, this seasonality is apparently related to reproductive activity (Buckworth and Clarke 2001). Observed seasonal variations in availability of Spanish mackerel probably reflect cycles of seasonal aggregation and dispersal (Buckworth *et al* 2007). For example, on the east and west coasts a portion of the Spanish mackerel stock migrate to the southern extent of their usual range during Austral Summer/Autumn months (Welch *et al* 2002; Tobin and Mapleston 2004). Between-sex differences in dispersal rates is evident, with males likely to be the most active dispersers (Buckworth *et al* 2007; Ovenden *et al* 2007).

Predators and prey
Prey of Spanish mackerel are mostly smaller fish (including other *S. commerson*), and squids (Kumaran 1964; Rao 1964; McPherson 1987). Fisher anecdotes also report that penaids are often a dominate gut item (A. Tobin pers. comm.).

Recruitment
Juvenile *S. commerson* are known to recruit to nearshore environments including estuaries (Jenkins *et al* 1984). From examples of age structure population 'snapshots' and from the very few time series of age structure data available, there is evidence of variable strength in year classes among years. Tobin & Mapleston (2004) were first to identify this characteristic in Spanish mackerel from the Queensland east coast and continued age structure sampling by Fisheries Queensland (QDAFF) has confirmed this trait. Variable annual larval growth and survival is to be expected given natural variation in environmental conditions and primary production. The collection of more robust time

series of age structure samples would inform more about recruitment variability. Inter-annual variation in catches of Spanish mackerel also suggests recruitment variation, though the quality of logbook data may compromise catch related analyses.

Current impacts of climate change
There are no documented impacts of climate change on Spanish mackerel.

Sensitivity to change

> **Key points:**
> - The sensitivity of Spanish mackerel to environmental variation is not well investigated, however annual spawning and southerly migrations on the east and west coasts appear to be temperature driven.
> - Juveniles live in estuarine and nearshore waters during the Austral wet season and so rainfall/riverflow, and associated primary production, are likely to be important drivers of juvenile survival.

The sensitivity of Spanish mackerel to changes in the environment is poorly understood, although temperature has been postulated to strongly influence spawning seasonality and annual migrations. The East Australian Current has been shown to have increased in strength and has extended further southward over the past 60 years (Ridgeway and Hill 2009) and this may have influenced the annual southerly migrations of Spanish mackerel into NSW during this time but has not been investigated. With future predicted warming of the oceans Spanish mackerel may become a far more important species in NSW with the possibility of increasingly southwards migrations and/or an increasing presence through range shifting.

Given that Spanish mackerel settle as juveniles in inshore and estuarine nursery areas (McPherson 1981a; Williams and O'Brien 1998), their early survival and growth are potentially influenced by local rainfall and river flows as has been documented for other species with inshore early life history stages (eg. Halliday *et al* 2008, 2011).

Resilience to change

> **Key points:**
> - Spanish mackerel are highly mobile, highly fecund and fast growing making them generally resilient. They also have a wide distribution across arnge of habitats. The availability and abundance of baitfish as prey may be the key limiting factor.

Spanish mackerel have a broad distribution range covering the entire northern Australian coastline. This, and the fact they are a highly mobile pelagic species, suggests they may be resilient to changes in the environment. They are also a highly productive and fast growing species, which makes them resilient to relatively high levels of fishing pressure.

Other

> **Key points:**
> - Spanish mackerel are known carriers of ciguatera. The effects of climate change on the incidence of ciguatera are poorly known but may be significant.
> - Research on the importance of temperature and riverflow (rainfall) as population drivers needs to be conducted.
> - Research on the sensitivity of early life history stages to altered sea temperatures and pH should also be conducted.

Ecosystem level interactions

Spanish mackerel are known carriers of ciguatera. Ciguatera poisoning is caused by a microscopic organism that attaches itself to algae growing in the warm waters of coral reefs. Small fish eat the algae, and are in turn eaten by larger fish such as Spanish mackerel. This food chain effect means that larger fish can accumulate enough of this organism to make their flesh toxic to humans when eaten. Ciguatera poisoning has been noted to be particularly prevalent in areas that have experienced some form of ecosystem disruption. Some examples of this may be pollution from industry, agricultural and human effluent, reef damage from cyclones, or coral bleaching triggered by rising water temperatures through the insidious effects of climate change. However, not all damaged reef environments exhibit outbreaks of ciguatoxic fish (Lewis and King 1996).

Additional (multiple) stressors

While harvest levels appear sustainable for most Spanish mackerel fisheries (Holmes *et al* 2012) there has been historical evidence of unsustainable harvest levels (see Begg *et al* 2006; Grubert *et al* 2013) suggesting this species is sensitive to poorly regulated fishing pressure.

Given the juvenile preference for nearshore waters, the survival of annual cohorts may be more affected by land-based influences on estuarine and nearshore conditions, such as changes in water quality and runoff that may negatively impact preferred habitats and/or prey.

Critical data gaps and level of uncertainty

Better estimates of recreational harvest levels for all stocks across northern Australia are needed given the importance of Spanish mackerel to this sector and the fully fished status of all fisheries. The sensitivity of Spanish mackerel early life history and adult stages to increases in temperature should be investigated since the spawning times and seasonal movements appear to be linked to certain times and places. The effect of rainfall/river flows on early life history survival and subsequent recruitment to the fishery should be investigated via recruitment indices and commercial catch data.

Acknowledgements

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23.   Spotted mackerel, *Scomberomorus munroi*

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The Australian spotted mackerel, *Scomberomorus munroi*, (hereafter referred to as the spotted mackerel) is a member of the family Scombridae (the tunas and mackerels). The species is distributed in the northern half of Australia, from the Abrolhos Islands in Western Australia (WA) to central New South Wales (NSW), and southern Papua New Guinea. The spotted mackerel is particularly significant to both recreational and commercial fishers in the Queensland East Coast Inshore Fin Fish Fishery (ECIFFF).

The fisheries

> - Catches in Western Australia, Northern Territory and the Gulf of Carpentaria are poorly known but are likely to be relatively low.
> - The majority of the catch is from the Queensland east coast but significant commercial and recreational catch is also made in NSW.
> - Commercial catch on the Qld east coast is regulated by a total allowable catch quota however, recreational catch in this fishery is not known.

Western Australia and Northern Territory

Commercial catches of spotted mackerel are negligible in Western Australia and are reported under "other mackerel" at 0.9 t in 2010 (WA Government 2011). There is no reported commercial catch of spotted mackerel in the Northern Territory however it is most likely taken and mixed in with other mackerel. In both Northern Territory and Western Australia most spotted mackerel landings are likely to be from the recreational fishing sector.

Queensland

In the Queensland Gulf of Carpentaria Inshore Fin Fish Fishery the commercial catch of spotted mackerel is low. In 2007, the only year for which data can be reported due to confidentiality reasons (less than 5 vessels), only 4 t was landed (DEEDI 2011a). The commercial catch in the Queensland Gulf of Carpentaria Line Fishery is not known but is unlikely to be high as the total commercial catch of all mackerel was 185 t in 2009 and was almost entirely comprised of Spanish mackerel (*Scomberomorus commerson*) with only 0.2 t of by-product species (DEEDI 2011b). The recreational catch in the Gulf of Carpentaria is not known. Spotted mackerel are landed in the Torres Strait Finfish Fishery but are reported under by-product species which is a minor component of the total catch (Marton et al. 2010).

The Queensland ECIFFF is a multi-species fishery comprising of charter, commercial, Indigenous and recreational fishing sectors. It is Queensland's largest and most diverse fishery. Fishing methods used are hook and lines (all sectors) and nets (commercial). Until 2002/03, commercial ring netting of spotted mackerel was legal and commercial catch was considerably larger than current levels (Figure 23.1). As of 2003/04, commercial landing of spotted mackerel in the ECIFFF has been

restricted to a total allowable catch of 140 t and the combined catch for all sectors is recommended not to exceed 296 t (DEEDI 2011b). Commercial landings were 100 t in 2009-10 and have been highly variable over the past four years (average annual catch: 65 t). Charter landings were 11 t in 2009/10 (DEEDI 2011b). The most recent estimate of recreational harvest from the statewide RFISH diary surveys was 148 t in 2005 (DEEDI 2008). Begg *et al*. (2005) re-analysed all RFISH data in a standardised manner, and obtained estimates of total annual recreational catch from Queensland of between 52 t and 265 t (mean of 175 t).
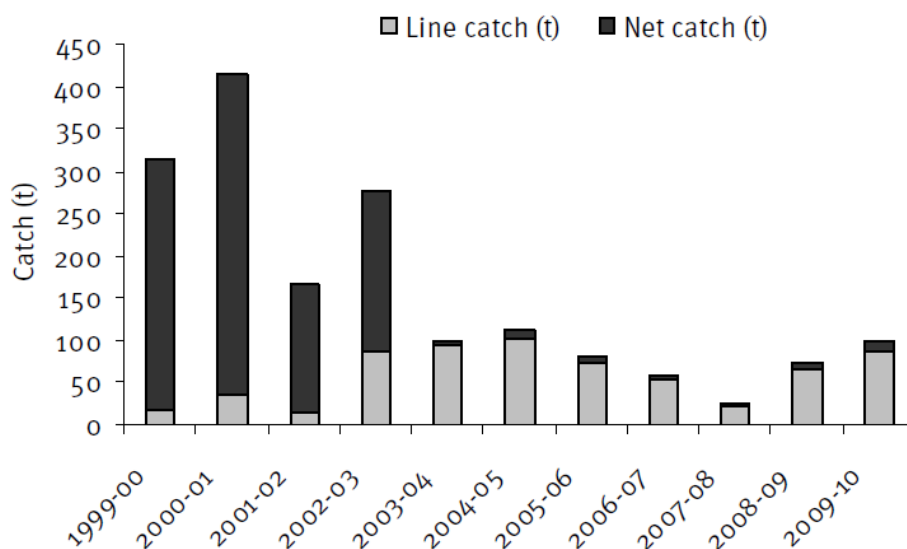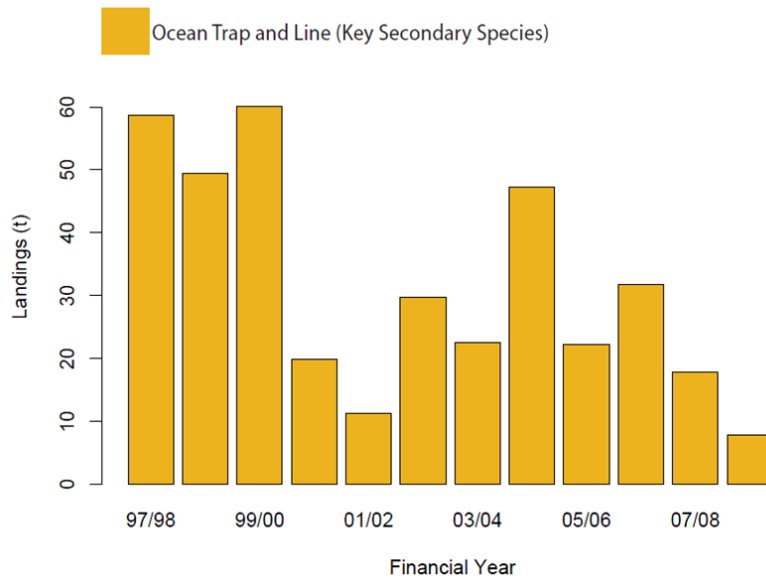

Figure 23.1. Commercial catch (t) of spotted mackerel caught by line and net in the ECIFFF, reported in logbooks 1999-00 to 2009-10. The substantial net catch prior to 2003-04 was the result of ring netting, which was banned from 2003-04 (Figure extracted from DEEDI 2011b).

Monitoring of spotted mackerel catches on the east coast since 2008 indicate that the commercial sector tend to take more fish in smaller size classes relative to the recreational sector. Very few fish over 95 cm TL are taken in either sector. The age structure taken by each sector is very similar and the majority of the catch by both sectors are comprised of 1- and 2-year old fish with fish older than 5 yrs being rare. The oldest fish sampled during monitoring was an 8-year old male (http://www.dpi.qld.gov.au/28_21410.htm, accessed 12/04/2012).

New South Wales
Spotted mackerel are landed commercially in the ocean trap and line fisheries in New South Wales as a key secondary species. The annual commercial catch ranges from less than 10 t to nearly 60 t (Figure 23.2). The species is also landed in the NSW recreational fishery and is thought to be between 10 t and 100 t (Rowling et al. 2010).

Figure 23.2. Reported landings of spotted mackerel by NSW commercial fisheries from 1997/98. Fisheries which contribute less than 2.5% of the landing are excluded (Figure extracted from Rowling et al. 2010).

Life history

- Life history knowledge for spotted mackerel is based entirely from research on east coast fish, which appear to be a single stock separate from the rest of Australia.
- They grow fast and mature early. Juveniles recruite to estuarine and nearshore areas.
- Adults move south into NSW during the austral summer and north for spawning from August to October in the Townsville/Mackay regions.

Life cycle, age and growth

A comprehensive understanding of spotted mackerel ecology and life history has been documented for the Australian east coast (Begg and Hopper, 1997; Begg et al., 1997; Begg, 1998; Begg et al., 1998; Begg and Sellin, 1998) (Figure 23.3). Spotted mackerel are dioecious (ie. separate sexes). Males attain sexual maturity between 401-450 mm FL and females between 451 to 500 mm FL and spawning occurs in northern Queensland waters from August to October (Begg, 1998). There is evidence that spawning is restricted to the waters between Townsville and Mackay (Jenkins et al. 1985). Aggregations of spotted mackerel seasonally occur mid-year north of Townsville, but these are not considered to be aggregations associated with any spawning (Cameron and Begg, pers. comm.). A tagging study has provided some evidence for a seasonal migration of spotted mackerel in that recaptures occurred to the north of the release sites (Rockhampton and Hervey Bay) in August and September but to the south of these locations during the Austral summer (Begg et al., 1997). Movements into NSW are therefore seasonal and restricted to the Austral summer and autumn months. Movements north of Cairns and in the Northern Territory and Western Australian waters are unknown. Known movements support a genetic study which identified that spotted mackerel form a single stock on the east coast of Australia and that this stock was genetically

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different from that in the Northern Territory (Cameron & Begg 2002). No information is available on Western Australian stocks or where the boundary between the Queensland east coast stock and the Northern Territory stock occurs.

Growth of spotted mackerel has been studied by Begg and Sellin (1998) for Queensland and NSW. In that study Von Bertalanffy growth parameters differ between the sexes but regional differences were minor with respective parameters in the range of: L_∞ = 727 to 729 mm FL, K = 0.272 to 0.339 and t_o = -4.00 to –2.53 for males and L_∞ = 823 to 866 mm FL, K = 0.41 to 0.52 t_o = -1.96 to -1.36. Longevity of spotted mackerel is at least 8 years and they are known to attain 104 cm and 10 kg in weight (Allen, 2009).
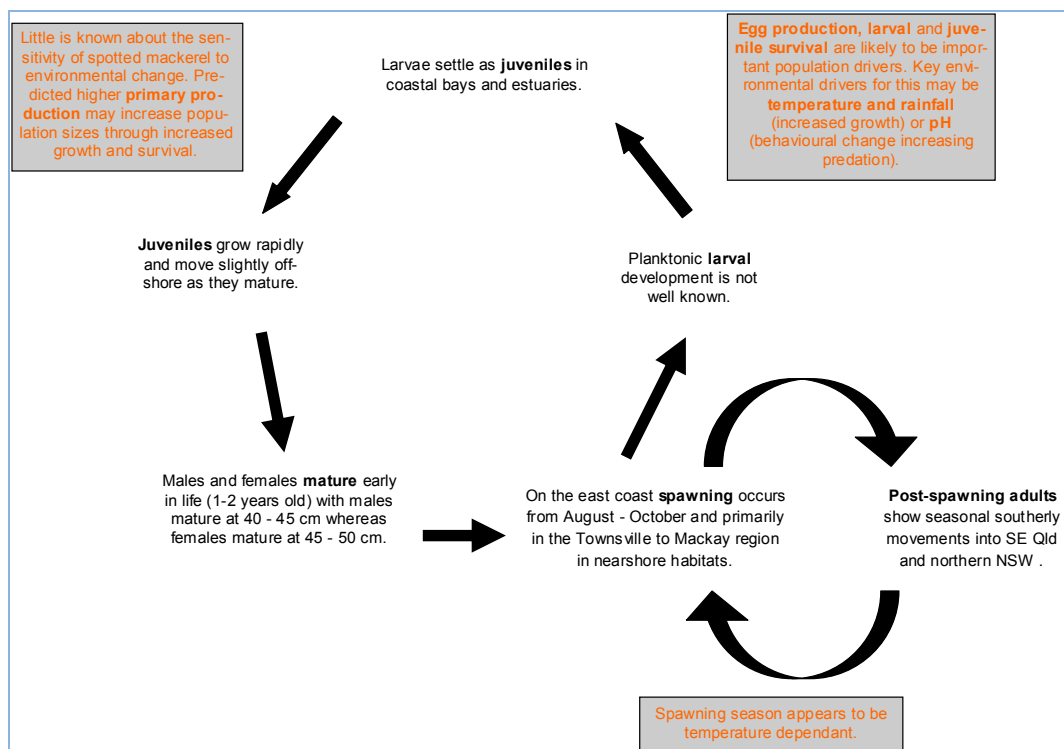

Figure 23.3. Generalised life cycle of the spotted mackerel, *S.munroi*, and the stages of potential environmental driver impacts. Life cycle is based on published research for the east coast stock (Begg and Hopper, 1997; Begg et al., 1997; Begg, 1998; Begg et al., 1998; Begg and Sellin, 1998).

Distribution, habitat and environmental preferences

The spotted mackerel is distributed in the northern half of Australia, from the Abrolhos Islands in Western Australia to central New South Wales, and southern Papua New Guinea (Figure 23.4). The species is found in coastal seas throughout the region. Research encompassing otolith microchemistry, genetic diversity, tagging and reproductive biology as well as seasonal variation in commercial harvesting strongly support the hypothesis that spotted mackerel form a single east coast stock with an annual large scale movement along the Queensland east coast to northern New South Wales. This includes Queensland and New South Wales feeding grounds in summer and a return migration in winter to northern spawning grounds (Begg and Hopper 1997, Begg et al 1997; Begg et al 1998; Begg 1998; Cameron & Begg 2002).
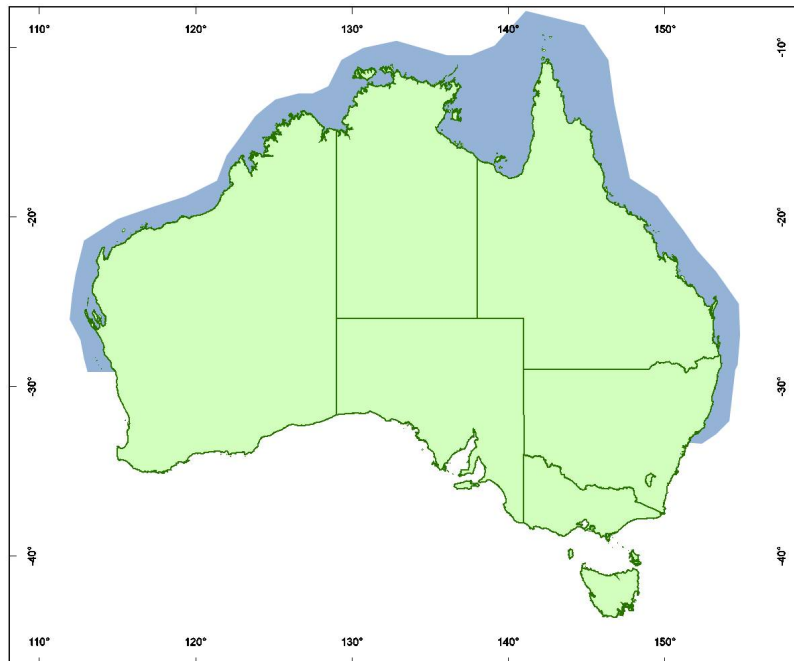
Figure 23.4. Australian distribution of spotted mackerel.

Predators and prey

Spotted mackerel feed primarily on fish, particularly Clupeids and Engraulids. The diet is also supplemented with invertebrates such as prawns and squid (Begg & Hopper, 1997).

Recruitment

There is no information on egg and larval development of spotted mackerel. They are known to have a pelagic larval phase and it is believed that larvae and juveniles move into coastal embayments and estuaries. This suggests that survival of annual cohorts are likely to be influenced not only by local environmental conditions, such as water temperature, salinity, pH, rainfall and river flow, but also by land-based influences on estuarine and nearshore conditions. Fishery independent measures of recruitment are unavailable across regions however, inter-annual variation in landings of spotted mackerel suggest some recruitment variation.

Current impacts of climate change

There are no reported impacts of climate change on spotted mackerel.

Sensitivity to change

- The known east coast spawning area is spatially and temporally restricted, i.e. north Qld from August to October, and likely to be linked to SST.
- Potential southerly extension of the species range and annual migratory pattern with increasing SST.
- Annual population recruitment may be influenced by local riverflow/rainfall since juveniles reside in estuaries and associated habitats.

As drivers for spawning are unknown, there is the potential for climate change to impact on spotted mackerel spawning patterns and distribution. Thermal tolerances are not understood however the species migrations appears to be correlated with water temperatures. The presence of reproductively active fish in waters of north Queensland is associated with a period of lower temperatures and annual southward migrations are correlated with warming waters during the Austral summer and autumn. The East Australia Current has been shown to have increased in strength and has extended further southward over the past 60 years (Ridgeway and Hill, 2009) and this may have influenced the annual southerly migrations of spotted mackerel into New South Wales during this time but this has not been documented anywhere. If, as is predicted, this trend continues, spotted mackerel may become a far more important species in New South Wales, especially since larger spotted mackerel are more typical in New South Wales, with the possibility of increasingly southwards migrations and/or an increasing presence through a range shift.

The early life history stages of most organisms are generally more sensitive to environmental conditions. Although not documented for spotted mackerel, based on evidence for other similar *Scomberomorus* species (see McPherson, 1978, 1981; Williams & O'Brien, 1998; Halliday et al, 2001), they may settle as juveniles in inshore and estuarine nursery areas. This makes early survival and growth potentially influenced by local rainfall and river flows as has been documented for other species with inshore early life history stages (eg. Halliday et al, 2008, 2011).

Resilience to change

- Spotted mackerel have a broad distribution across northern Australia, are highly mobile and highly productive.
- Annual spawning (on the east coast) is restricted in time and place.

Spotted mackerel have a broad distribution range covering the entire northern Australian coastline. Although at least two stocks are apparent across this range, the spatial scale of stock division is vast. For example, the east coast population of spotted mackerel, at least to Cairns in the north, has been shown to represent a single stock (Begg et al., 1997; Begg et al., 1998; Begg and Sellin, 1998). This broad distribution, and the fact they are a highly mobile pelagic species, suggests they are resilient to changes in the environment. Further, spotted mackerel have been shown to have variable growth rates, as well as size and age at maturity and sex change depending on location and possibly population densities indicating they are an adaptable species to varying environmental and population conditions. They have a broad diet with two of their major prey families (Clupeids and Engraulids) some of the most prolific baitfish species throughout northern Australia.

Other

- Recreational harvest levels are poorly understood.
- A close association of early life history stages with inshore estuarine habitats suggests rainfall/riverflow and land-based influences may affect population recruitment, however this is poorly understood.

Ecosystem level interactions

Spotted mackerel rely on schooling baitfish as prey species and the effects of climate change on baitfish species remains very poorly understood.

Additional (multiple) stressors

Spotted mackerel are a popular target species on the Australian east coast, particularly for recreational fishers, which will inevitably increase with increasing human populations. The commercial fishery was historically heavily fished, however management changes that introduced a total allowable commercial catch (TACC) and restricted commercial fishing gear to hook and line has reduced annual catch from a peak of 410 t in 2000-01 to the average annual catch over the past four years of 65 t (DEEDI, 2011b). The level of recreational catch is poorly understood.

Given the likelihood of juvenile preference for nearshore waters the survival of annual cohorts may be more prone to being affected by land-based influences on estuarine and nearshore conditions, such as changes in water quality.

Critical data gaps and level of uncertainty

Better estimates of recreational harvest levels for both Queensland and New South Wales need to be determined given their importance to this sector and the high level of uncertainty in current estimates. Future assessments should use data from each jurisdiction (Qld, NSW) since the east coast is assumed to represent a single stock.

The sensitivity of spotted mackerel early life history and adult stages to increases in temperature and rainfall should be investigated since the spawning areas appear to be linked to certain times and places (north Qld waters during August-October), and the fishery sector allocation implications of a southerly shift in their range. The effect of rainfall/river flows on early life history survival and subsequent recruitment to the fishery should be investigated via recruitment indices and commercial catch data.

Acknowledgements

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Implications of climate change impacts on fisheries resources of northern Australia

Part 2: Species profiles