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Assessing the water needs of fisheries and ecological values in the Gulf of Carpentaria

Final Report prepared for the Queensland Department of Natural Resources and Mines (DNRM)

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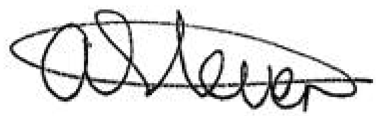
Foreword

The recent CSIRO Flinders and Gilbert Agricultural Resource Assessment (FGARA) report evaluates the feasibility, economic viability and sustainability of agricultural development in the Flinders and Gilbert catchments in the south-eastern Gulf of Carpentaria, and demonstrated the potential for developing their water resources and arable soils. However, the rivers flowing into the coastal waters of the Gulf of Carpentaria currently support commercial fisheries, particularly for Barramundi and Banana Prawns, recreational and Indigenous fisheries and key ecological values ranging from estuary and river-floodplain habitats to endangered Largetooth Sawfish. Hence, whilst the demand for irrigated agriculture in northern Australia is rapidly increasing so too is the competition for water between different uses. Regional assessments that address ecological, cultural and socio-economic values therefore need to be undertaken in parallel to agricultural assessments so that tradeoffs between competing demands for water can be effectively explored. Such assessments will provide foundations for prioritisation, decision-making and implementation of water resource management plans for the future development of northern Australia.

There is clear recognition and obligation under the Queensland Government Gulf Water Resource Plan (WRP) that there should be a balance between future agricultural development and the ecosystem services supported by existing river flows. The aim of this collaborative project between CSIRO and the Queensland Department of Natural Resources and Mines is, therefore, to assess the potential risks to fisheries and ecological values in the southern Gulf of Carpentaria from the range of FGARA development scenarios. A key finding of this assessment is that water management strategies designed to mitigate the environmental and economic impacts of changed flows need to ensure that flows and connectivity between marine, estuarine and freshwater ecosystems are sufficient to support species and habitats identified at high risk, and to ensure the maintenance of flow-dependent fishery catches. The assessment reported here will aid policy makers and water resource managers anticipate and address water trade-offs from catchment to coasts.



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Shortened forms

AWRC	Australian Water Resources Council
BBN	Bayesian Belief Network
CLCAC	Carpentaria Land Council Aboriginal Corporation
CPUE	Catch Per Unit Effort
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DAFF	Queensland Department of Agriculture, Fisheries and Forestry
DNRM	Queensland Department of Natural Resources and Mines
DSITIA	Queensland Department of Science, Information, Technology, Innovation and the Arts
ENSO	El Niño-Southern Oscillation
FGARA	Flinders and Gilbert Agricultural Resource Assessment
GL	Gigalitre
GoC	Gulf of Carpentaria
Gulf	Gulf of Carpentaria
GWRP	Gulf Water Resource Plan
ML	Megalitre
NASY	Northern Australia Sustainable Yields
NCRM	Natural and Cultural Resource Management
NGNRM	Northern Gulf Natural Resource Management
NPF	Northern Prawn Fishery
NQIAS	North Queensland Irrigated Agriculture Strategy
NRM	Natural Resource Management
NT	Northern Territory
GOCIFF	Gulf of Carpentaria Inshore Finfish Fishery
QGF	Queensland Gulf Fishery (the GOCIFF plus the Mud Crab fishery)
QRA	Quantitative risk assessment
Pdf	Probability distribution function
PDO	Pacific Decadal Oscillation
SGC	Southern Gulf Catchments
t	Tonnes
TOR	Terms of Reference
TRaCK	Tropical Rivers and Coastal Knowledge Research Hub
WRP	Water Resource Plan
WYQ	Wet Year Flow volume (ML, October to September)
YCS	Year Class Strength

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Executive summary

1 The estuaries and rivers of the south-eastern Gulf of Carpentaria (GoC) region support many species that are culturally important (e.g. Turtles and Dugongs) and have high conservation value (e.g. sawfishes and shorebirds). There are also valuable Indigenous, recreational and commercial fisheries for a range of finfish species such as Barramundi and crustaceans such as the Mud Crab. Offshore White Banana Prawns are the target of a lucrative trawl fishery. These species all depend on the natural flow of the region's rivers and streams.

2 The CSIRO Flinders and Gilbert Agricultural Resource Assessment (FGARA) project was a comprehensive, integrated evaluation of the feasibility, economic viability and sustainability of agricultural development in the Flinders and Gilbert catchments in the GoC, and demonstrated the potential for developing their water resources and arable soils.

3 Nevertheless, there is clear recognition and obligation under the Gulf WRP that there should be a balance between future agricultural development and the ecosystem services supported by existing river flows. The aim of this project, therefore, is to assess the potential risks to fisheries and ecological values in the GoC from the range of FGARA development scenarios.

4 Risks to most species from sustained reductions in flow were assessed qualitatively because of time constraints and significant gaps in ecological knowledge related to flow dependence. The qualitative risk assessment approach used explicit criteria defined by Australian Standards to rank the consequence and likelihood of reduced flows affecting the abundance of each taxonomic group, and was addressed by extensive literature review of life histories, conceptual modelling, elicitation of expert knowledge and consultation with some key affected stakeholders. Whilst the approach cannot quantify the risk of reduced flow to the long-term viability of a species, it served the purpose of identifying and prioritising potential species of concern from highly diverse and data-limited ecosystems in the southern GoC.

5 A total of 46 taxonomic groups and four habitats were identified as being potentially at risk from reduced flow in the Flinders and Gilbert rivers. Of this vulnerable group, 15 (33%) were assessed as being at high risk, 8 (17%) at moderate risk, 13 (28%) at low risk and 10 (22%) at negligible risk. Mangrove and seagrass habitats were assessed at low risk but coastal floodplains and salt flats were assessed at moderate risk.

6 The species of lowest risk generally completed most of their life cycles in offshore waters and had wide geographic distributions of single populations that extended beyond the GoC. Species of highest risk shared several common traits. For example, many fish species were highly euryhaline, using coastal waters to reproduce as adults, with juveniles moving into estuarine and freshwater habitats. Such life history traits make populations of these species highly vulnerable to reduced river flows through reduced connectivity of key habitats and availability of prey.

7 Detailed quantitative risk assessments of reduced flow in the Flinders and Gilbert rivers were undertaken for White Banana Prawns and Barramundi given their socio-economic importance, their well established strong dependence on freshwater flows to maintain populations, and the availability of long-term fishery catch and effort data. The assessment end point used for both species was catch weight, and is a strong indicator of the economic performance of the fisheries.

8 Statistical models were developed to predict Banana Prawn catch from fishing effort and flow from the Flinders and Gilbert rivers combined. These were used to simulate 'what if' scenarios that encompassed the range of water harvests associated with FGARA development proposals (i.e. 2-13% of the mean long-term wet year flow or 2-20% of the median).

9 Results overall predict that water harvests will reduce Banana Prawn catch in adjoining fishing zones by 3-13%. However, this is likely an overestimate given that four other major rivers also contribute to total flow in southern GoC waters. Fine-scale spatial modelling is required to tease out the degree of over-estimation.

10 The FGARA development scenarios entail water allocations above current entitlements (scenario A) for the Flinders and Gilbert catchments and provide a range of estimates for reduced flow to estuarine and marine ecosystems, both as a percentage of the mean and median (Flinders: 1-20% of the mean wet year flow and 3-41% of the median; Gilbert: 1-20% and 1-28% of the mean and median, respectively.

11 Barramundi catch was also predicted from fishing effort and flow but separately for the Flinders and Gilbert rivers. A 3-12% reduction in catch is predicted for Gilbert FGARA development scenarios and a reduction of 2-4% for the Flinders. The results for the Flinders should be treated with caution, however, as most (90%) observed catches are explained by fishing effort compared to flow, reflecting a slowly recovering fishery from decadal-scale droughts during the 1980s concomitant with heavy fishing.

12 The results of the 'what if' assessments for Banana Prawns and Barramundi fisheries are similar: the proposed FGARA development scenarios will unlikely lead to the imminent collapse of either fishery, although some level of reduced catch will occur. However, whilst the predicted reductions in catch for both fisheries may seem moderate, they could have greater effect in combination with other cumulative risks to the fisheries not addressed here (e.g. reduced water quality associated with agriculture, loss of floodplain habitat from sea level rise and the effects of prolonged droughts).

13 The risk to Barramundi population recruitment from FGARA development scenarios was quantified also using Year Class Strength (YCS) analysis. Results for both rivers show that no years were classified as high risk for Barramundi populations under pre-development and current water use scenarios. In contrast, under full use water scenarios, the high risk category increased from 0 to 3% in the Flinders River and from 0 to 12% in the Gilbert River.

14 The catch-flow and YCS results are consistent in that they both demonstrate that the relative effects of reduced flow are four times greater in the Gilbert than the Flinders. However, under full use water development scenarios the moderate YCS risk category increased from 10 to 24% in the Flinders River and from 5 to 26% in the Gilbert River. In contrast to the catch-flow model results, this suggests that the long-term sustainability of the Barramundi fishery could be at risk unless mitigation strategies are adopted (see Part 4).

15 The most important caveat is that our model results cannot be validated in the real world and so should be treated with caution. They can only be used as a guide to possible future scenarios, not as prescriptions of exactly what will occur because of the underlying assumptions and inherent uncertainties in our correlative catch-effort-flow model. The only way to increase certainty is to increase systems knowledge, which requires investment in underpinning science.

16 We used the results of both the qualitative and quantitative assessments to develop potential mitigation strategies so that decision makers may make more informed trade-offs between different water uses. That is, given the underlying risks identified, what water allocation strategies will increase the likelihood that assets are maintained and/or risks minimised?

17 The attributes of those species and habitats evaluated at high and moderate levels of risk from reduced flow were first summarised to help identify patterns in the underlying exposure and effects pathways, in order to evaluate the effectiveness of potential mitigation strategies for as many species or species groups as possible. Reduced flow past some unquantified threshold will reduce seasonal connectivity between rivers and their floodplains due to decreases in the extent, frequency and duration of floods. This will in turn reduce productivity in these habitats and, as a result, the reproduction, survival and growth of flow-dependent species will be reduced.

18 Hence, mitigation strategies require that flows and connectivity are sufficient to support high risk species and habitats, and to ensure the maintenance of fishery catches. Knowledge of fine-scale dynamics of some species, such as White Banana Prawns and Barramundi, may require mitigation strategies at finer temporal scales than annual flow targets.

19 Mitigation strategies should be implemented in an adaptive management framework complete with a monitoring program to ensure that such strategies are effective. A strategic research plan should be implemented to close critical knowledge gaps and comprise an integral component of any mitigation strategy. For example, lack of ecosystem-level understanding of the effects of development could be addressed by ecosystem modelling (see Part 2, Recommendations), which would help frame policy alternatives as testable hypotheses linked to monitoring. Additionally, population-level understanding of key species in the Gulf of Carpentaria Inshore Finfish Fishery is required to better anticipate, mitigate and monitor potential impacts of agricultural development.

20 The full economic impact of sustained reductions in fisheries catch is not addressed here (see Part 5, Section 22) but is a critical part of any assessment of water tradeoffs between agricultural and fisheries production. Additionally, economic and non-market impacts of water extraction on Indigenous and recreational catch, and on conservation values, were also not part of this assessment (Part 5, Section 23). We therefore recommend strongly that studies be implemented to address these serious shortcomings, particularly with respect to Indigenous ecological values of coastal and marine resources that may be impacted by sustained reductions in flow.

21 Further consultation and engagement with key stakeholders (e.g. the NPF and the Gulf of Carpentaria Commercial Fishermen's Association; Indigenous, recreational and conservation interests; farmers and graziers) is required to communicate our risk assessments accurately.

22 To conclude we highlight the fact that the FGARA development scenarios that underpin this assessment will, at the end of the day, likely comprise only one of many alterantive development pathways for these catchments. The focus of our assessment, however, has been on understanding the potential changes to current 'End of System' flow patterns regardless of the details of how development proceeds in the catchments, and to assess what risks this may pose to ecological values and fisheries production in the marine-estuarine environment of the south-eastern Gulf of Carpentaria.

Part 1 Introduction

Rik Buckworth, Cuan Petheram, Rob Kenyon and Peter Bayliss

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1 Background

There is considerable impetus to develop Australia's north. The annual wet season rains of Australia's tropical north are seen as a potential source of water for irrigation for agriculture, as well as town water and other industrial uses. For instance, the Land and Water Taskforce (2009) envisaged that there could be a 40% increase in value of production (GVP) from agricultural production in northern Australia between 2000 and 2030. At the same time, this vision was tempered by recognition that this development needs to be sustainable, that water available for agriculture, as well as arable soils, is limited, and that water **flowing to the ocean** is not 'going to waste'.

The Water Resource (Gulf) Plan (2007) (Gulf WRP; *Queensland Water Act 2000*) states that water is to be allocated and sustainably managed. It seeks a balance in the development of water use that recognises social and cultural needs and aspirations of communities, as well as supporting fishing and tourism (e.g. by protecting natural aesthetics of watercourses) and that water should be available to support natural ecosystem processes. The Gulf WRP details several general ecological outcomes which are to be maintained, including natural variability in flows, connectivity – allowing for the movements of animals – and the delivery of water, sediment, nutrients and organic matter through the system.

The south-eastern Gulf of Carpentaria (GoC) region is of importance to tourism and culture. The estuaries and rivers of the region support numerous species that are culturally important (e.g. turtles) and have conservation value (e.g. sawfishes, shorebirds). There are valuable Indigenous, recreational and commercial fisheries for a range of finfish species and crustaceans: iconic species such as Barramundi and Mud Crabs are fished in the estuaries and near-shore areas. Further offshore, the White Banana Prawn is the target of a lucrative trawl fishery. All these species are dependent, in various ways, on the flow of the region's rivers and streams.

The Flinders and Gilbert Agricultural Resource Assessment (FGARA) project (Petheram et al. 2013d,e; http://www.csiro.au/fgara; see Section 2 below) was a comprehensive and integrated evaluation of the feasibility, economic viability and sustainability of agricultural development in the Flinders and Gilbert catchments in north Queensland. It demonstrated the potential for developing the catchments' water resources and arable soils. At the same time, there is recognition and obligation under the Gulf WRP, that balance should be achieved between development of those resources and the services supported by existing flows.

This project aims to deliver a technical assessment of potential impacts to fisheries and ecological values in the GOC from a range of development scenarios examined in the FGARA study. The project focuses on impacts in the marine and estuarine areas of the catchments.

The project proceeded by identifying the 'important' ecological assets – species or groups of species or places (see Gulf WRP) – in the estuarine and marine environments connected to the receiving waters of the Flinders and Gilbert rivers. The freshwater component of the Department of Natural Resources and Mines (DNRM) Gulf Assessment is being addressed by another project. The project team engaged with a reference group of interested stakeholders, invited the input of experts in various taxa or systems, and searched the literature to elicit what might be 'important' ecological assets.

Risks to important ecological assets were evaluated qualitatively using likelihood–consequence approaches. For two of the species, Barramundi and White Banana Prawn, there was sufficient information from the fisheries to undertake more detailed quantitative risk analyses.

We evaluated how mitigation might be achieved. We asked 'what would be required of water allocation scenarios to ensure that these assets were maintained and/or risks minimised?', so that decision makers might better assess trade-offs between different uses.

The reference group and experts provided excellent input into evaluating risks and discussion of mitigation ideas. Nevertheless, the scope of the project meant that we were unable to engage with all stakeholders or experts who might have provided further input on the importance of particular species or other ecological assets.

For most species, risks assessments were restricted to qualitative analyses. Most quantitative analyses and in-depth modelling were simply beyond the scope of the project. But additionally, the significant knowledge gaps for most species precluded in-depth analysis. Consequently, a key feature of this report is identifying additional work that should or might be undertaken to support effective decision making.

2 The Flinders and Gilbert Agricultural Resource Assessment: an overview of hydrology

2.1 Scenario definitions

The Flinders and Gilbert Agricultural Resource Assessment (FGARA) considered two hydrological scenarios when evaluating climate, surface water, groundwater and economic development (Petheram et al. 2013a,b):

- Scenario A – historical climate and current development
- Scenario B – historical climate and future irrigation development

Future climate scenarios were not addressed.

Scenario A

Scenario A included historical climate and current development. The historical climate data cover 121 years (water years from 1 July 1890 to 30 June 2011) of observed climate (rainfall, temperature and potential evaporation for water years). All results presented in this report cover this period unless otherwise specified. 'Current development' is the current level of surface water, groundwater and economic development that was defined as at 1 July 2013. FGARA assumes that all current entitlements are being used in full. Scenario A was the baseline against which assessments of relative change were made. Historical tidal data were used to specify downstream boundary conditions for flood modelling undertaken by FGARA.

Scenario B

Scenario B included historical climate and future irrigation development, elaborated by FGARA through discussion with stakeholders. Scenario B used the same historical climate data as Scenario A. Future irrigation development is described by each case study storyline, and river inflow and agricultural productivity were modified accordingly.

2.2 Case studies

FGARA considered three case studies in the Flinders catchment. Their purpose is to evaluate the scale of opportunity for irrigation in key geographic areas of the catchment. By analysing water storage options and potential crops, they enable assessments of the viability and sustainability of irrigated agriculture.

Three case studies were undertaken in the Flinders catchment:

1. Cave Hill Dam and irrigated sorghum (grain) (Flinders Scenario B.1, Cloncurry River, 40-GL annual yield, maximum storage capacity 248-GL; see Petheram et al. 2013a for details)
2. O'Connell Creek off-stream storage and irrigated rice (Flinders River Scenario B.2, Flinders River, 34-GL annual yield, 127-GL maximum storage capacity; see Kim et al. 2013 for details)
3. Irrigation mosaics and a variety of irrigated crops (Scenario B.3; see Holz et al. 2013 for details)

Three case studies were undertaken in the Gilbert catchment:

4. Green Hills Dam and irrigated three-crop rotation (Gilbert Scenario B.1; 172-GL annual yield, 227-GL maximum storage capacity; see Hughes et al. 2013 for details)
5. Dagworth and Green Hills Dams and irrigated sugarcane (Gilbert Scenario B.2; Dagworth/Einasleigh River, 326-GL annual yield, 498-GL maximum storage capacity; see Poulton et al. 2013 for details)

Kidston Dam and irrigated Rhodes grass (Brennan et al. 2013 for details).

As part of the FGARA, the case studies that resulted in the largest perturbations to flow in each catchment were examined in greatest detail. These were the irrigation mosaics and a variety of irrigated crops at the largest water extraction in the Flinders catchment, and the Dagworth and Green Hills Dams and irrigated sugarcane in the Gilbert catchment. These two case studies are discussed in more detail below.

Flinders water harvesting

This case study investigated water harvesting along the Flinders, Cloncurry, Corella and Woolgar rivers in the Flinders catchment that supplies water for small irrigation developments on individual properties. Water harvesting refers to extracting water during flow events and either applying it directly to a crop, or more commonly, holding it in offstream storage on a property for later use (Figure 1.1).

The rationale for using offstream storages is that potential large instream dam sites in the Flinders catchment generally have small water yields and are situated a long way from areas of suitable soil. In many cases, there is considerable uncertainty about their geological foundations. The Flinders catchment does, however, have soils that are likely to be suitable for small offstream storages (Figure 1.1), with 68% of the catchment underlain by clay soils.

In this case study, it is assumed that farms, particularly offstream storages, are supplied by pumps. The larger the pump capacity, the more frequently an irrigator can extract their annual entitlement. Despite larger pump capacities having larger capital and ongoing operation and maintenance costs, it was found that it would be more economically viable for individual property owners to have large pump capacities. As a result, a storage size to entitlement-to-pump capacity' ratio of five was adopted for this case study. This means that, if pumps are operating at full capacity, it takes five days to fill the offstream storage.

Water would be applied to a field using spray irrigation or best-practice surface irrigation, which includes laser levelling and on-farm tail-water recycling. As a result, it is assumed that very little tail-water runoff occurs (i.e. water leaving the field following an irrigation event), except during large rainfall events, which may occur immediately after irrigation on full soil profiles.

This case study examined five new catchment entitlements (Flinders Scenario B.3):

- 80-GL catchment entitlement – corresponding to Scenario B80
- 160-GL catchment entitlement – corresponding to Scenario B160
- 240-GL catchment entitlement – corresponding to Scenario B240
- 400-GL catchment entitlement – corresponding to Scenario B400
- 560-GL catchment entitlement – corresponding to Scenario B560.

These new entitlements are in addition to the existing entitlement, that is, in addition to the 80 GL of water that was released in 2013 and to the pre-2013 entitlement of around 25 GL.

We note here that whilst the FGARA development scenarios outlined above underpin our risk assessment, in reality they only comprise one of many possible alterantive development pathways for these catchments. The focus of our assessment, however, is to understand the potential changes to current 'End of System' flow patterns regardless of the details of how development proceeds in these catchments and, accordingly, to assess what risks this may pose to ecological values and fisheries production in the marine-estuarine environment of the south-eastern Gulf of Carpentaria.

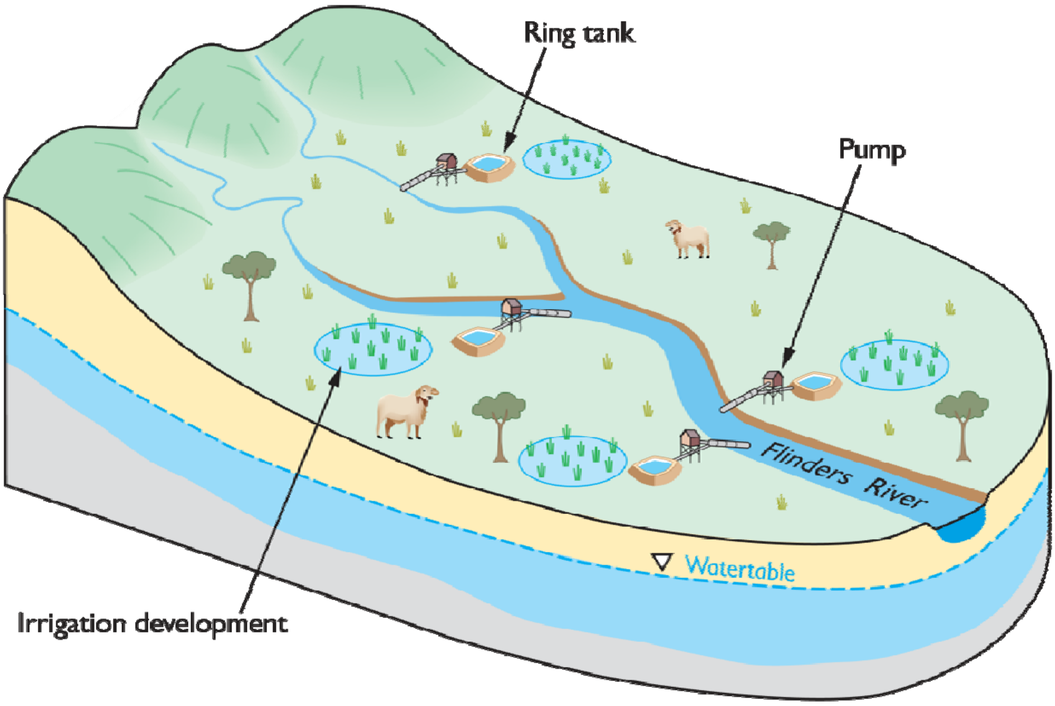

Figure 1.1 Schematic diagram of the components in the case study on water harvesting irrigation development in the Flinders catchment.

Dagworth and Green Hills dams and irrigated sugarcane

This case study investigates the potential of two irrigation developments involving two dams on the Gilbert and Einasleigh rivers. The dams and associated infrastructure would supply irrigation water to 15,000 ha of sugarcane adjacent to each of the Gilbert and Einasleigh rivers (Figure 1.2). The potential Green Hills (Gilbert River) and Dagworth (Einasleigh River) dams are 20 m and 30 m high, roller-compacted concrete dams, respectively. The developments would enable sugarcane to be supplied to a newly established sugar mill in the Gilbert catchment.

Approximately 20 km downstream of the potential Green Hills Dam a re-regulating structure (sheet-piling weir) would be constructed. As it is unlikely that rock foundations would be present, it is assumed that a 325-m wide, 3-m high sheet-piling weir would need to be constructed. On the Einasleigh River, sand dams would be constructed approximately 70 km downstream of the potential Dagworth Dam. These sand dams are low embankments comprising riverbed sands that partially span the lower Einasleigh River. They are constructed downstream of a natural waterhole to form a deep pool from which to pump water. Although sand dams are cheap to construct, compared with a concrete or sheet-piling weir, they have much larger seepage losses beneath and through the dam wall, and need to be rebuilt every year.

Both potential irrigation developments are situated 2 km from the river due to the presence of marginally suitable land in the vicinity of the river. This enables a 2-km wide riparian zone to be maintained between the irrigation development and the river. It is assumed that irrigation water is applied to the field using modern spray irrigation systems capable of delivering peak water requirements to the cane crop at periods of high-evaporative demand. Well-managed spray irrigation generates very little tail-water runoff, except during large rainfall events that may occur immediately after irrigation on full soil profiles.

See Holz et al. (2013) for a comprehensive description of the case study.

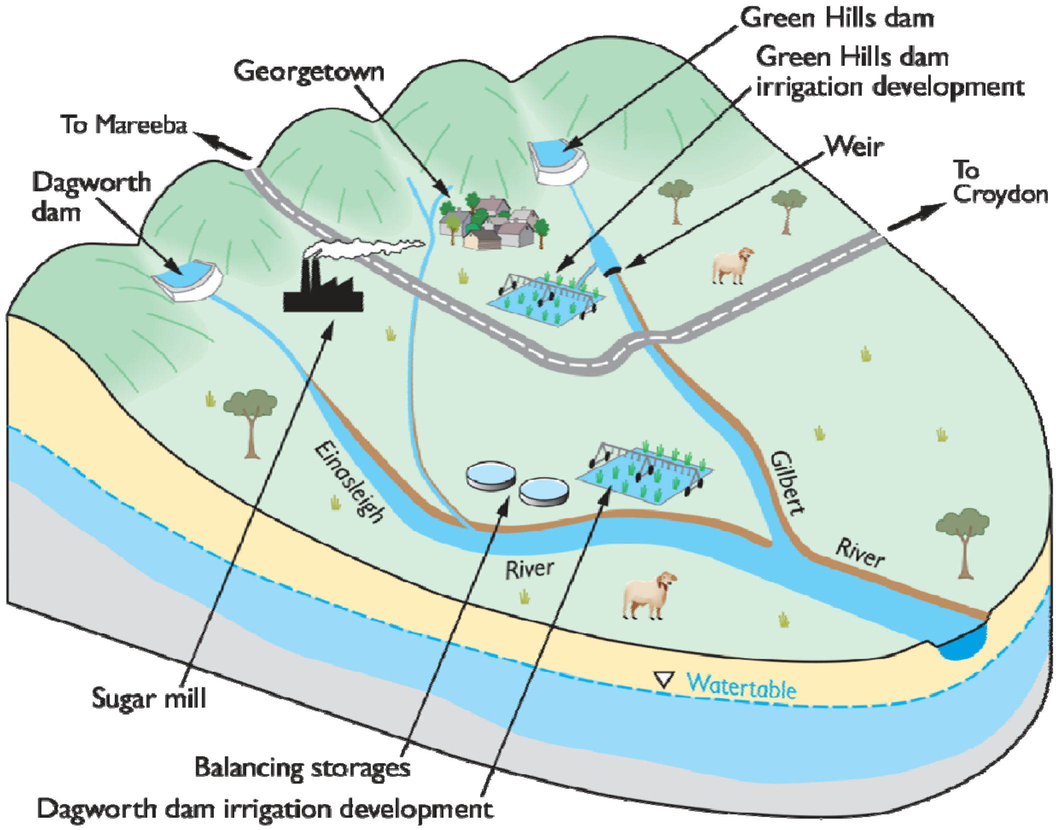

Figure 1.2 Schematic diagram of the components in the case study on the irrigation developments associated with Green Hills and Dagworth dams.

2.3 Hydrology

This section provides an overview of the hydrology of the Flinders and Gilbert catchments. The material was summarised from relevant information presented in Petheram et al. (2013b,c).

The Flinders River is the main river of the Flinders catchment and is the longest river in Queensland and sixth longest river in Australia. It has a mean and median annual flow at its most downstream gauge of 2543 GL and 1241 GL, respectively. It rises in the Great Dividing Range, 100 km north-east of Hughenden (Figure 1.3). The river flows from north to south, until it reaches Hughenden where it tracks west across flat and naturally treeless Mitchell grass plains. After flowing through the town of Richmond, it continues towards the north-west before flowing north and draining into the GOC. Unlike many rivers in northern Australia, the Flinders River has a good quality streamflow gauging station in its lowermost reaches.

Two major rivers comprise the Gilbert catchment, the Gilbert and the Einasleigh (Figure 1.3). At the confluence of the Gilbert and Einasleigh rivers, the mean annual streamflow is about 3706 GL. Due to a couple of very wet years elevating the mean, this amount of water is more than 40% greater than the median annual streamflow (2585 GL). Although the Gilbert catchment is named for the Gilbert River (named after the explorer Gilbert), the median annual streamflow in the Einasleigh River is about two-and-a-half times that of the Gilbert River. The Gilbert and Einasleigh rivers converge at Strathmore Station before forming a river delta 100-km wide and then flowing into the GOC. There are no gauging stations below the confluence of the Gilbert and Einasleigh rivers. This makes runoff and streamflow estimates in the lower part of the catchment very uncertain.

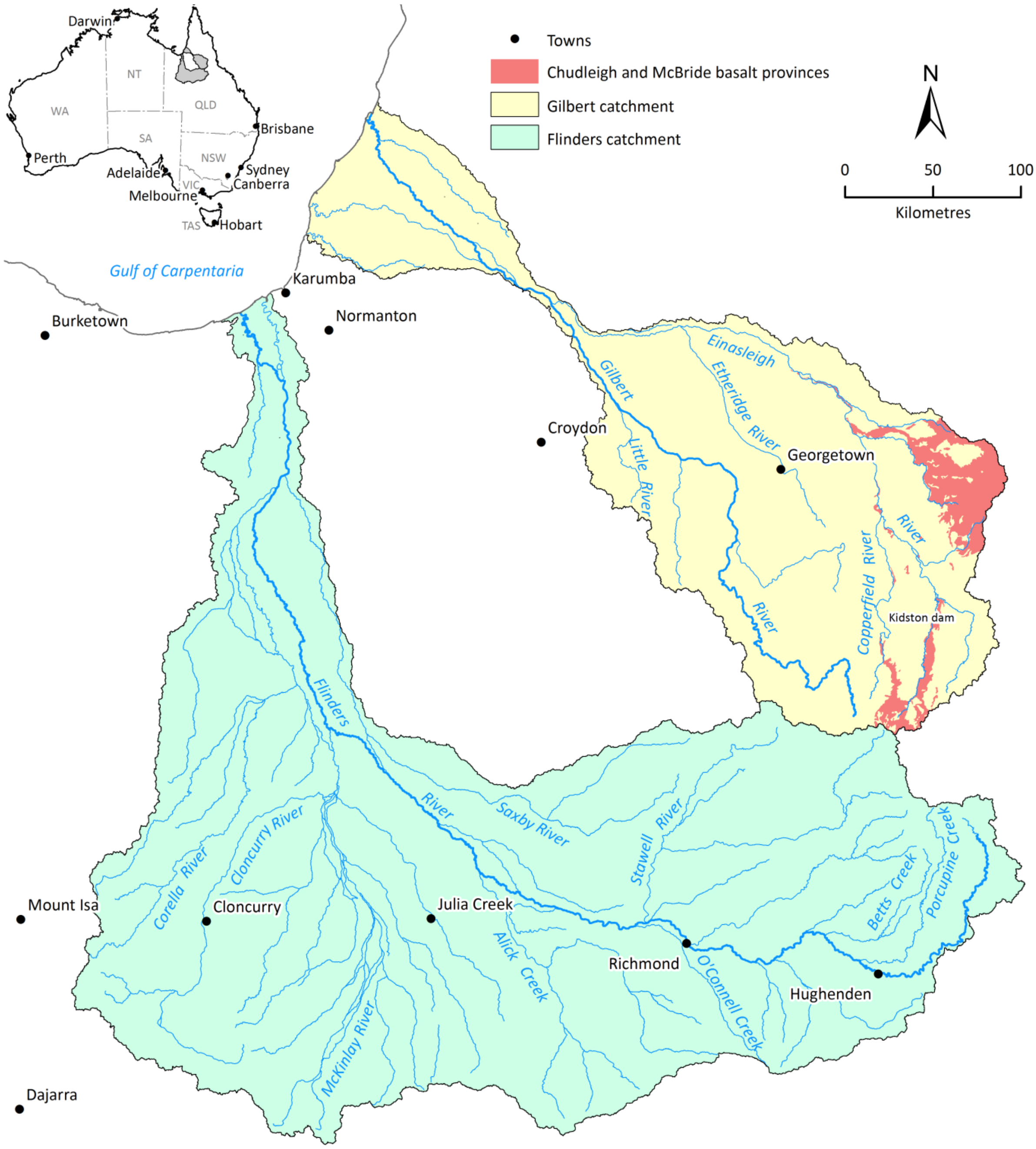

Figure 1.3 Major rivers and towns in the Flinders and Gilbert catchments.

The Flinders and Gilbert catchments are two of 25 Australian Water Resources Council (AWRC) catchments that drain into the GoC (Figure 1.4). Separated by the Norman catchment, they are flanked by the Staaten and Mitchell rivers to the north of the Gilbert, and the Leichhardt, Nicholson and other western GoC catchments to the west of the Flinders. The surrounding catchments also contribute significant flows to the marine receiving waters of the GoC. End of system (EOS) flows from the Gilbert and Flinders rivers are comparable to the flows from these adjacent rivers. Preliminary analyses indicate that the proportional contribution of EOS from the rivers of the south eastern GoC are: Flinders 16%; Gilbert 24%; Leichhardt 11%; Norman 22% and Staaten 26%. The hydrology of the Flinders and Gilbert catchments and the relationship that these have to Banana Prawn and Barramundi catches are further examined in Part 3.

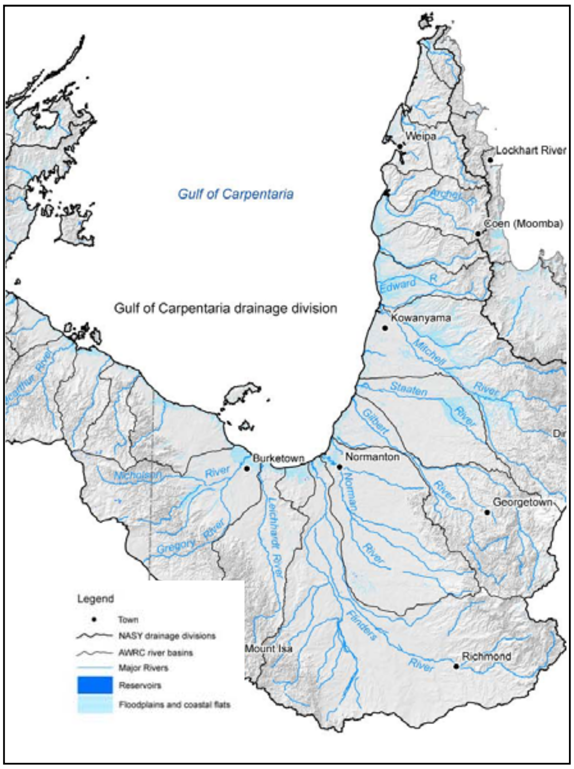

Figure 1.4. Catchments (AWRC river basins) in the Gulf of Carpentaria (modified from Fig. 3 in Petheram et al. 2009a).

Intra and inter-annual variability

Runoff and streamflow in the Flinders and Gilbert catchments are highly variable within and between years. More than 95% of all runoff in the catchments occurs during the wet season, which is very high compared with rivers in southern Australia (Petheram et al., 2008). As a result, most rivers in the Flinders and Gilbert catchments are ephemeral. Most of the streamflow gauging stations record cease-to-flow conditions more than 80% of the time for the Flinders, and more 50% for the Gilbert. Notable exceptions are the tributaries draining the Chudleigh and McBride Basalt provinces in the eastern parts of the Gilbert catchment, which flow well into the dry season or are perennial.

Once streamflow ceases in these catchments at the end of the wet season, the rivers break up into a series of waterholes during the dry season. Many of these waterholes gradually disappear in time as water is lost through evaporation and seepage. Some, however, persist throughout the dry season. Waterholes that persist from one year to the next provide key aquatic refugia that are considered ecologically important for sustaining ecosystems.

The coefficient of variation (a measure of variability) of annual runoff in the Flinders and Gilbert catchments is 1.2 and 1.1, respectively. Based on data from Petheram et al. (2008), the variability in runoff in the Flinders catchment is comparable to the annual variability in runoff of other rivers in northern and southern Australia with a comparable mean annual runoff. It is, however, two to three times more variable than rivers from the rest of the world of the same climate type as northern Australia (Petheram et al. 2008).

Flooding

Flood events provide an opportunity for offstream wetlands to be connected to the main river channel, which is important for aquatic animals to achieve completion of important life-cycle stages. The high biodiversity found in many unregulated floodplain systems in northern Australia is thought to largely depend on flood events, which allow for biophysical exchanges to occur between the main river channel and wetlands.

The coastal floodplains of the Flinders catchment (Figure 1.5) are particularly susceptible to flooding and flood events and can extend many hundreds of kilometres inland. This is because a number of rivers draining large areas of land (e.g. Flinders, Gilliat, Cloncurry and Corella rivers) are funnelled into an area about 50 to 100 km wide. Furthermore, the mid- to lower reaches of the Flinders catchment are very flat, so floodwater drains slowly. A consequence of this low relief is that during large flood events, water can cross between the Flinders and Norman catchments to the north-west.

The coastal floodplains of the Gilbert catchment (Figure 1.5) are not as susceptible to flooding as those in the Flinders catchment, nonetheless flooding may extend many tens of kilometres inland. Above the confluence of the Gilbert and Einasleigh rivers, regular widespread flooding does not occur.

See Poulton et al. (2013) for a comprehensive description of the case study.

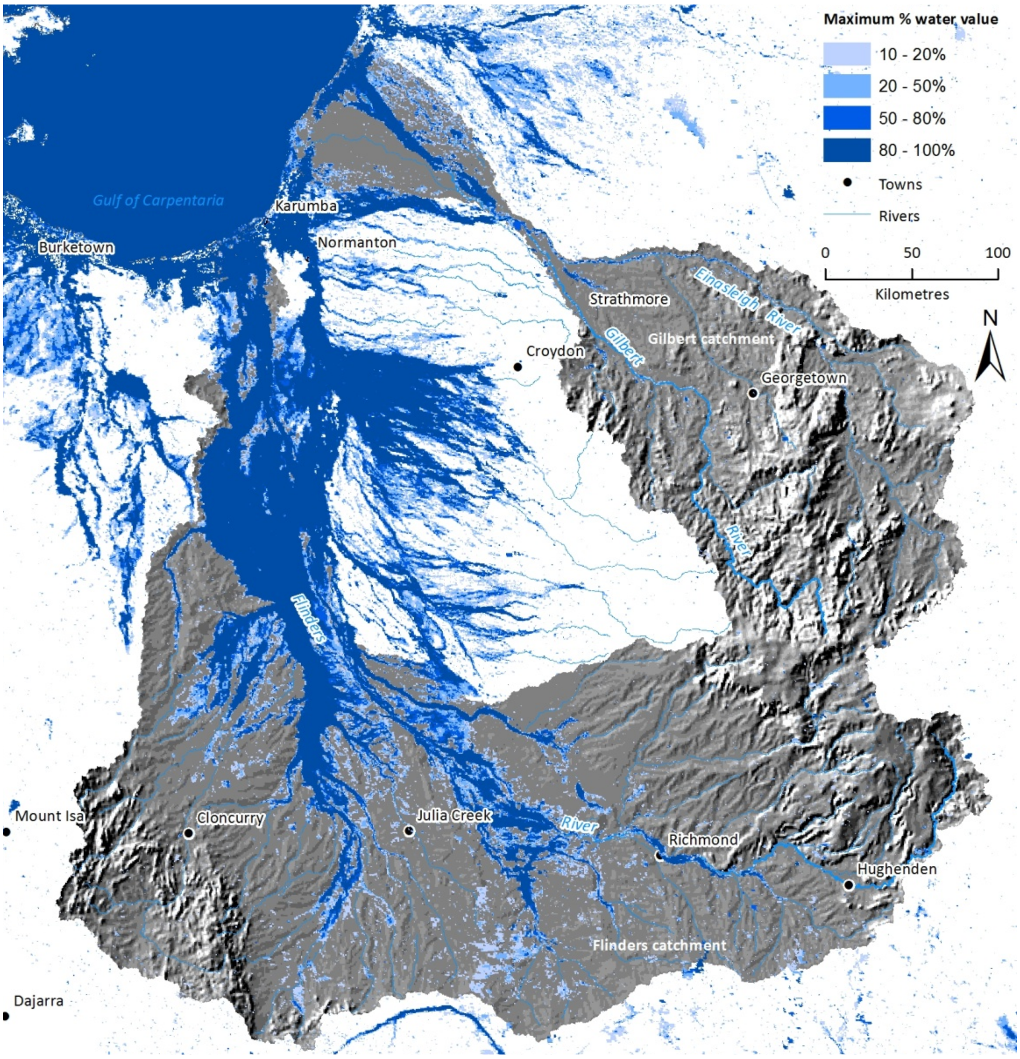

Figure 1.5 MODIS [1] flood map generated from Maximum OWL [2] flood maps from the relatively wet years of 2000, 2001, 2004, 2008, 2009, 2010 and 2011 with artefacts removed. Sourced from Dutta et al. 2013.

1 Moderate Resolution Imaging Spectroradiometer

2 Open Water Likelihood

3 Objectives

The overarching requirement of this project was to provide an analysis of the risk of possible impacts to fisheries and ecological values in the GOC, from key development scenarios specified in FGARA. The specific objectives requested by DNRM were to:

1. Identify relevant species within the Gulf fishery (commercial and non-commercial) and any relevant species of high ecological importance (e.g. only exists in the area; endangered; keystone species etc.) that are dependent on flows from the Flinders and Gilbert rivers.

We achieved this objective by indentifying relevant species and detailed their life histories using a range of approaches. We reviewed the literature and mined existing datasets to compile a species list of teleosts, sharks and rays, reptiles and invertebrates for the Flinders and Gilbert rivers and the adjacent estuarine and coastal habitats. Sources included a range of journal publications and literature describing field surveys undertaken by the CSIRO, Queensland Government agencies and universities, and databases compiled from various surveys in northern Australia as well as a database developed for a Commonwealth Ecological Risk Assessment project for the Northern Prawn Fishery. Two workshops (Appendix 1) were held with the project team, members of a reference group of interested stakeholders, and an expert panel, including scientists with knowledge of key fishery species and their flow dependencies. This enabled us to further elicit species or assets of importance to fisheries, their conservation status, their potential importance as a keystone species for maintaining the integrity of the ecosystem, or their endemism to the region.

2. Assess the importance of contributions made by flows from the Flinders and Gilbert rivers to fisheries and ecological values in the greater Gulf of Carpentaria region.

This objective was achieved. Information gathered in the review process included fisheries and other datasets. We also reviewed the significant body of research that describes the relationship between flows and fisheries and ecological values in the region. These supported both qualitative and quantitative analyses of the relationship between important species and flow.

3. Identify the flow dependencies for species identified in objective 1 including end-of-system flow requirements.

We achieved this by identifying and compiling life-history flow requirements for the important species identified in Objective 1. Based on this information, we developed conceptual models detailing important trophic, ecological, economic and social links relevant to each key species. Given these models, we assessed the importance of flows with respect to key fisheries values (e.g. catch, value) and ecological values (e.g. change in biomass, trophodynamics, spatial and temporal distributions, productivity). The project workshops were also used to further develop and examine the conceptual models. The knowledge and skills of the expert panel as well as reference group members provided input to identify likely flow dependencies for data-poor species. The workshop process enabled further expert input on flow dependencies.

4. Identify the relative risks/threats that development poses to fisheries and ecological values.

We completed this objective by applying qualitative risk assessment methods to the species that were identified as important (using the conceptual models). The approach used is consistent with national and international guidelines (e.g. AS/NZS 4360:2004, now superseded by AS/NZS ISO 31000:2009). Given the sound conceptual models developed as described above, we proceeded by ranking and prioritising development threats to fisheries and ecological values, developing a likelihood-by-consequence matrix for each species or asset given the FGARA scenarios. Again, we used the workshops to gain wide input on development of the likelihoods.

5. Assess the relative risk of possible impacts to fisheries and ecological values for development scenarios based on key scenarios identified under the FGARA.

In addition to the qualitative analyses described above, this objective was further addressed by quantitative analyses feasible for two of the more important and data-rich species: White Banana Prawns in the Northern Prawn Fishery, and Barramundi in the Queensland Gulf of Carpentaria Inshore Finfish Fishery. Relationships between catch and fishing effort and flow for these two species were modelled, providing quantitative predictions of the effects of altered flow regimes on the landings of these species in the commercial fisheries.

6. Recommend potential mitigation strategies to minimise the identified risks and evaluate the effectiveness of the proposed mitigation strategies.

This objective was achieved. Using life-history attributes and risks to the important species and other assets related to different temporal and spatial scales, we developed mitigation strategies that acknowledged these different scales. The project workshops were again important in providing contributions to these strategies.

7. Identify priority knowledge gaps, and monitoring and research needs to fill those gaps.

This objective was achieved. The information gathering and analytic exercises described led to the identification of spatial and temporal gaps in our knowledge. These would need to be filled before additional quantitative region- and species-specific or ecosystem-scale analyses might be undertaken to more thoroughly assess the potential impacts of altered flow regimes. Although we did extensive, quantitative analyses of banana prawns and barramundi, few other species considered here have sufficient data or knowledge available to inform such work.

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Part 2 Qualitative risk assessment of the potential effects of altered river flows in the Flinders and Gilbert catchments

Shane Griffiths, Rob Kenyon, Elvira Poloczanska, Roy Deng, David Milton, Peter Rothlisberg, Michele Burford, Frank Coman and Margaret Miller

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Summary

In this ecological risk assessment for the Flinders and Gilbert river catchment, available catch and effort data were collated for commercial, recreational and Indigenous fisheries as well as fishery independent aquatic faunal and trophic surveys in the southeastern GoC. This identified 46 taxonomic groups that were considered by scientific experts and GoC stakeholder groups as being potentially at risk of being negatively affected by reduced river flows. Unfortunately, the enormous complexity of the coastal, estuarine and freshwater ecosystems in the study region and the general paucity of quantitative biological and ecological data available for the vast majority of species — particularly rare or species of low commercial value that make up a significant component of the ecosystem (Blaber, 2000; 2002) — means that rapid assessment of the sustainability of each species potentially impacted by altered river flows is a difficult prospect. It is acknowledged that many hundreds of species may be affected by reduced flows, but given the time and resources available to the project it was simply not feasible for all species to be formally assessed.

A wide range of species are potentially impacted by changes in the flow characteristics of the Flinders and Gilbert rivers. In some cases, neither the species itself, nor the most conspicuous phase of its life history might directly use riverine or estuarine habitats. This is because the life-history strategy of some species does not use the impacted habitats; or their inconspicuous larval and juvenile phases may use riverine or estuarine habitats. However, some of these species support significant GoC fisheries. For example, the White Banana Prawn supports a 50-vessel fishing fleet in offshore waters of the GoC. However, the critical habitat for the juvenile phase of this species is mangrove forests and creeks within estuaries, where flow characteristics are vitally important to its growth and emigration. A further example is that many coastal shark species may not use or rely on estuaries at any stage, yet the populations of their principal prey species in the lower estuary and near-shore habitats rely on the health of coastal habitats that are sustained by floodplume nutrient deposits. As a consequence, predators such as sharks and other predators may be indirectly impacted by altered flows.

In contrast, some species such as the threatened largetooth sawfish are prominent as adults in nearshore and offshore coastal zones, and as juveniles and adolescents in the freshwater habitats. Longstream and lateral connectivity is therefore crucial to their life success. Moreover, Brown and Grooved Tiger Prawns have a near-identical life history to White Banana Prawns, yet they are likely to be little impacted by reduced flows. This is because their juvenile habitat is located in embayment and shallow-coastal seagrass communities rather than estuarine mangrove habitats; a critical difference between the co-occurring species. Consequently, we have provided life history summaries for each species or species-group to enable a full understanding of each facet of the life history of the species and how key life history stages might be impacted by altered flows.

The qualitative risk assessment approach employed in this project used explicit criteria defined by Australian Standards to rank the consequence and likelihood of reduced flows affecting the population size and dynamics of each taxonomic group by way of expert opinion and stakeholder consultation. Although the approach does not quantify the risk of a perturbation to the long-term sustainability of a species, it is particularly useful for identifying and prioritising potential species of concern from highly diverse and data-limited ecosystems such as the Flinders and Gilbert rivers and the adjacent coastal zone. In general, the species of lowest risk completed most of their life cycle outside of estuaries in offshore waters and generally had wide geographic distributions that represented a single population that extended beyond the GoC. As a result, the effects of a change in river flows due to irrigation on these species were considered unlikely to be detectable against natural background variability in their population dynamics.

In contrast, the species of highest risk shared several common traits. Many of these fish species were highly euryhaline and used coastal waters to reproduce as adults and moved, long distances in some cases, into estuarine and freshwater habitats as nursery areas for juveniles. Interestingly, many species have evolved a specialised reproductive strategy of being protandrous hermaphrodites, which may ensure the sustainability of their populations in highly variable environments of the wet-dry tropics. Furthermore, several species have highly subdivided populations (e.g. Blue and King Threadfin, Pikey Bream, Barramundi), to the extent where individual rivers in the southeastern GoC most likely support self sustaining populations of some species. Together, such life history traits make the population size and dynamics of these species highly vulnerable to being negatively impacted by reduced river flows through reduced connectivity of key habitats and availability of prey.

It is important to note that qualitative risk assessments, including the approach used here, can only provide a relative measure of risk from the defined perturbation. This means that the 16 highest risk species may in fact not be considered as being truly at risk without employing further quantitative population assessments in relation to reduced river flows. Furthermore, this assessment has not considered the cumulative impacts of other external pressures on these populations such as fishing, climate change, or the effects of increased turbidity, nutrification or contaminants that may result from agricultural development along the Flinders and Gilbert rivers. These will be important considerations in assessing the viability of populations in these rivers and require dedicated data collection programs to inform the single species and ecosystem models needed to quantitatively elucidate the impacts of specific perturbations.

5 Background to the risk assessment

5.1 Characterisation of the ecosystem, fisheries, species prioritisation and risk assessment

In order to assess the potential impacts of reduced river flows on individual species and ecosystem integrity, a stepwise process was used beginning from a description of the ecosystem and fisheries to assessing the risks of altered river flows on individual taxon. The first step in the process was to characterise the system by bringing together available information that described the faunal composition of the ecosystem. Because this project primarily focuses on the impact of reduced river flows on species of importance to fisheries, the spatial extent of the analysis was restricted to marine, coastal and estuarine habitats, and including some euryhaline species that use freshwater during part of their life cycle (e.g. Barramundi, Sawfishes). Freshwater species and habitats are being concurrently assessed by other state government departments in relation to reduced flows.

Data were derived from a wide range of sources including journal publications and grey literature describing field surveys, as well as unpublished data held by the CSIRO, and Fisheries Queensland. Most surveys of river, estuarine and marine habitats employ specific sampling methods to address the hypothesis being tested. Due to the significant variation in size and preferred habitats of particular species, a single sampling method is usually not adequate for describing an entire assemblage. Therefore, in order to characterise all trophic levels, we included data from dietary studies from the region and effectively used predators as 'biological samplers' of the ecosystem.

After collating available datasets, species were identified for closer investigation as part of a formal qualitative risk assessment. First, any threatened, endangered or protected (TEP) species listed under the *Environment Protection and Biodiversity Conservation (EPBC) Act 1999* or the International Union for the Conservation of Nature (IUCN) Red List were immediately identified and included in our final list of species to be assessed. Species listed as 'Least Concern' by the IUCN were not included as their populations have been formally assessed and considered sustainable.

Second, using fishery logbook catch data (expressed as weights) from Fisheries Queensland and the CSIRO, species were identified that were of commercial importance to the Northern Prawn Fishery (NPF) and Queensland's Inshore Fin Fish, Mud Crab, Line, and Developmental Fin Fish Trawl fisheries. Initially, the principal target species of each fishery were included, which was then supplemented with other byproduct species based on their rank in annual catches or on advice from stakeholders in Workshop 1.

Species of importance to the recreational fishery in the study region were identified using data for the Gulf of Carpentaria (GoC) from the 2010 Queensland statewide survey (Taylor et al., 2012). Species of importance to Indigenous fisheries were identified from the Queensland component of the National Recreational and Indigenous Fishing Survey (NRIFS) (Henry and Lyle, 2003).

An attempt was made to look more broadly at potential ecosystem impacts and to identify any keystone or indicator species that may play an important role in maintaining the structural integrity of the ecosystem. This was done by using the model of the GoC ecosystem built by Griffiths et al. (2010) to investigate the ecological effects of prawn trawling by the NPF. Although this model characterises the entire GoC ecosystem, it comprises many coastal and estuarine ecological functional groups, which may provide us with an indication of the relative importance of such groups to the overall functioning of the GoC system.

After identifying the key species and habitats, an ecological risk assessment determined the relative risk that the populations of individual species and habitats would become unsustainable in the long term under a scenario of reduced river flows in the Flinders and Gilbert rivers. For consistency and simplicity, common names and the convention for their capitalisation (e.g. Blue Threadfin) as defined by the Australian standard for common fish names (Yearsley et al., 2006) are used in this report.

5.2 Description of the fisheries

13.3.1 NET FISHERIES (N3, N11, N12 AND N13)

The Gulf of Carpentaria Inshore Fin Fish Fishery operates in the Queensland waters of the GoC that comprises the Flinders and Gilbert rivers and may be affected by changes in flows of these rivers. The fishery also contains four net fisheries (N3, N11, N12 and N13) as well as recreational and charter fisheries. The N11 is a small baitfish fishery and is not included in this report.

The N3 is an inshore gillnet fishery that operates within 7 nm of the Queensland mainland or island shore. The fishery employs set mesh gillnets to primarily target Barramundi, Blue Threadfin, King Threadfin and tropical sharks. In 2013, a total of 87 N3 symbols were held by 81 licences. Of the 81 licences endorsed to access the N3 fishery, 74 were reported as having actively fished.

The N12 and N13 are offshore gillnet fisheries that primarily target sharks and Grey Mackerel. The N12 and N13 extend from 7 nm and 25 nm from shore to the extent of the Australian Fishing Zone. In 2013, only three licences had access to the N12 fishery, and there were no licences issued for the N13 fishery (Queensland, 2014).

The three net fisheries are managed by a series of input controls (e.g. net length and mesh size restrictions, limited entry) and output controls (e.g. minimum and maximum size limits). Fishers are required to record catch (by species) and effort in a logbook. The fisheries were monitored by scientific observers to verify logbook data, but the program was discontinued in 2012. Although the fisheries are open year round, a closure on the take of all Barramundi between 7 October and 1 February effectively closes estuarine fishing in the N3 during this period.

Net fishing effort in the GoC has been relatively stable since the early 1990s to 2004 at around 11,100-13,000 days per year. After this time, effort decreased to a low of 7551 days in 2013 (Figure 2.1).

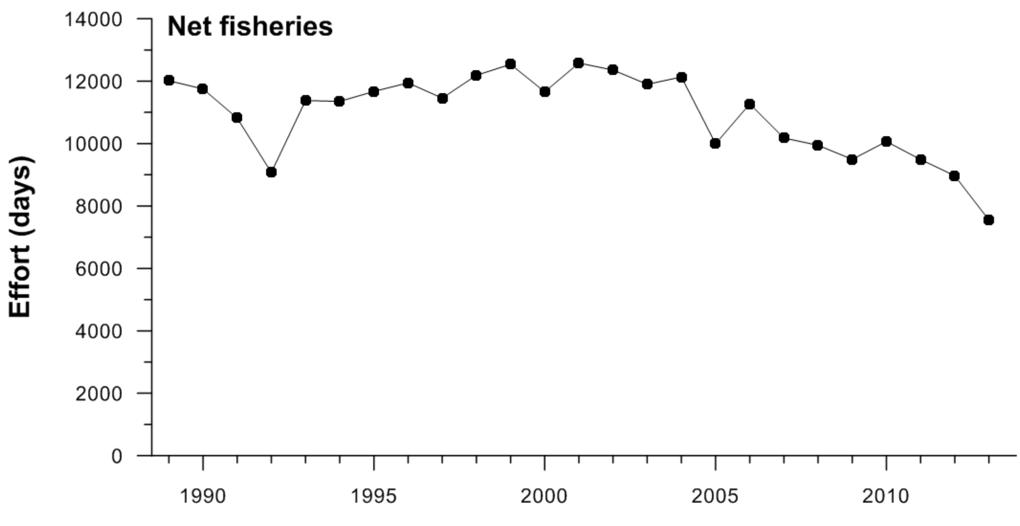

Figure 2.1. Annual fishing effort of net fisheries in the Gulf of Carpentaria, combined for the N3, N11, N12 and N13 fisheries. Data supplied by Fisheries Queensland.

13.3.2 MUD CRAB FISHERY (C1)

The C1 Mud Crab fishery (C1 symbol) operates in the GoC from the high tide mark to 7 nm from shore. Fishers holding a C1 symbol in the Gulf of Carpentaria almost exclusively exploit Mud Crabs (Scylla spp.) using baited traps and pots. Since Mud Crabs spend most of their life cycle within estuaries, this fishery may be affected by changes in river flows in the Flinders and Gilbert rivers.

The Mud Crab fishery is one of the largest fisheries in the GoC, with 62 licence holders reporting 6010 days of effort in 2013. The GoC Mud Crab fishery was valued at $2.8 million in 2013 (Queensland, 2013a).

The fishery is managed by a series of input controls (e.g. 50 pots per licence, limited entry) and output controls (e.g. minimum size limits, prohibition on taking female crabs). Fishers are required to record catch (by species) and effort in a logbook.

Annual fishing effort in the Mud Crab fishery in the GoC has steadily increased from 79 days in 1991 to a peak of 7535 days in 2003 (Figure 2.2). Effort has been relatively stable in the past five years averaging 6185 days per year.

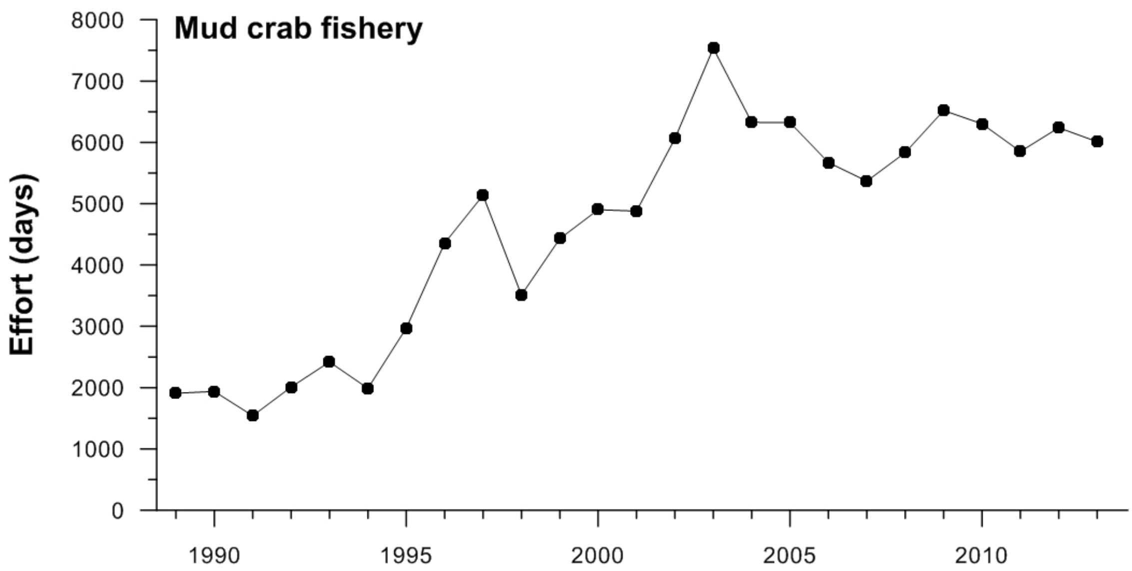

Figure 2.2. Annual fishing effort of the C1 Mud Crab fishery in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

13.3.3 LINE FISHERY (L4)

The GoC Line Fishery (L4 symbol) operates from shallow coastal waters to offshore regions of the GoC to the Northern Territory border. The principal target species is Spanish Mackerel, which is generally caught offshore using surface trolling. However, hand lines are also used at times to catch tropical Snappers and Emperors, some of which are associated with estuaries in their juvenile phases. Therefore, marketable species in this fishery may be affected by changes in river flows in the Flinders and Gilbert rivers.
A total of 46 licences held L4 fishing symbols in 2013, of these licences 15 reported catch, and was valued at $986,000 (Queensland, 2013a). The fishery is managed by a series of input controls (e.g. number of lines and hooks, limited entry) and output controls (e.g. minimum and maximum size and in-possession limits). Fishers are required to record catch (by species) and effort in a logbook. This fishery has a closed area extending south of the Mitchell River, effectively removing effort from waters under the direct influence of discharge from the Flinders and Gilbert rivers. The fishery is also prohibited from retaining Barramundi, Black Jewfish, Blue Threadfin, King Threadfin, Scaly Jewfish, Silver javelin and Talang Queenfish in order to reduce interactions with the N3 component of the Inshore Fin Fish Fishery.

Annual fishing effort in the GoC Line Fishery has been variable in the GoC, ranging between 1087 days in 1989 to peaks of 1733 days in 1997, with the most recent peak of 1562 in 2008. Since then, effort has decreased to 517 days in 2013 (Figure 2.3).

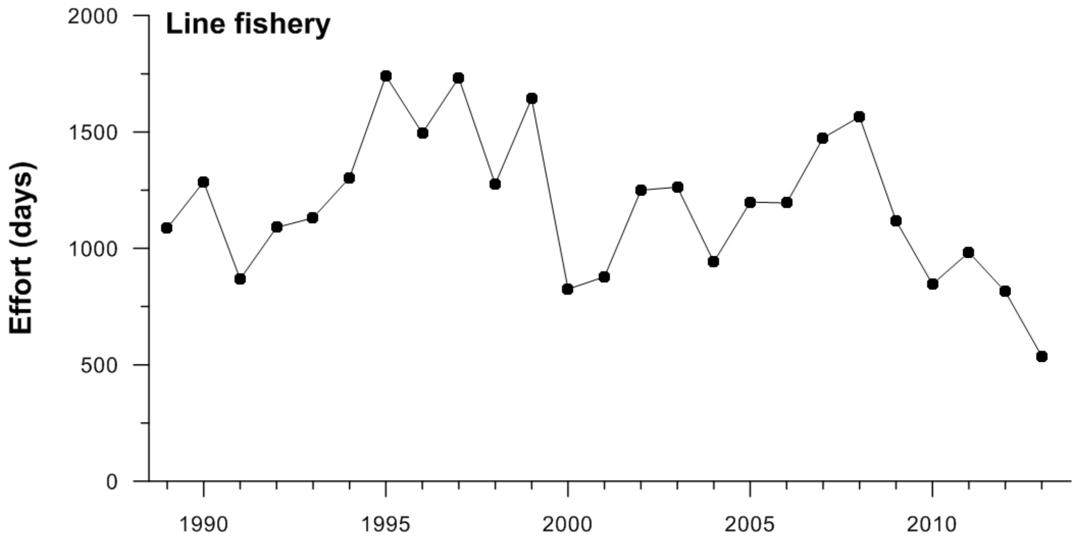

Figure 2.3. Annual fishing effort of the L4 line fishery in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

13.3.4 DEVELOPMENTAL FIN FISH TRAWL FISHERY

The Gulf of Carpentaria Developmental Fin Fish Trawl Fishery operates in offshore waters of the GoC, generally to the west of northern Cape York. Although the reef-associated species Crimson Snapper and Saddletail Snapper are the primary target species, the fishery also catches large quantities of Mangrove Jack and Golden Snapper, which occur in estuaries in their juvenile phases. As such, catches of target species in this fishery may be affected by changes in river flows in the Flinders and Gilbert rivers.

The fish trawl fishery is a limited-entry, quota managed, semi-demersal trawl fishery that has operated under the Queensland Fisheries Joint Authority since 1998. Given the developmental status of the fishery, no symbols have been issued, however, currently there are three permits being utilised by three licences. The total allowable commercial catch of nominated target species (i.e. Lutjanids) for this fishery is currently set at 1250 t, of which only 2% was used in 2013. In 2013, the fishery was valued at $173,000 (Queensland, 2013b).

Fishing effort increased significantly from 2 days in 2001 to 390 days in 2008; wheras in 2012 and 2013, effort was just 39 days and 7 days, respectively (Figure 2.4).

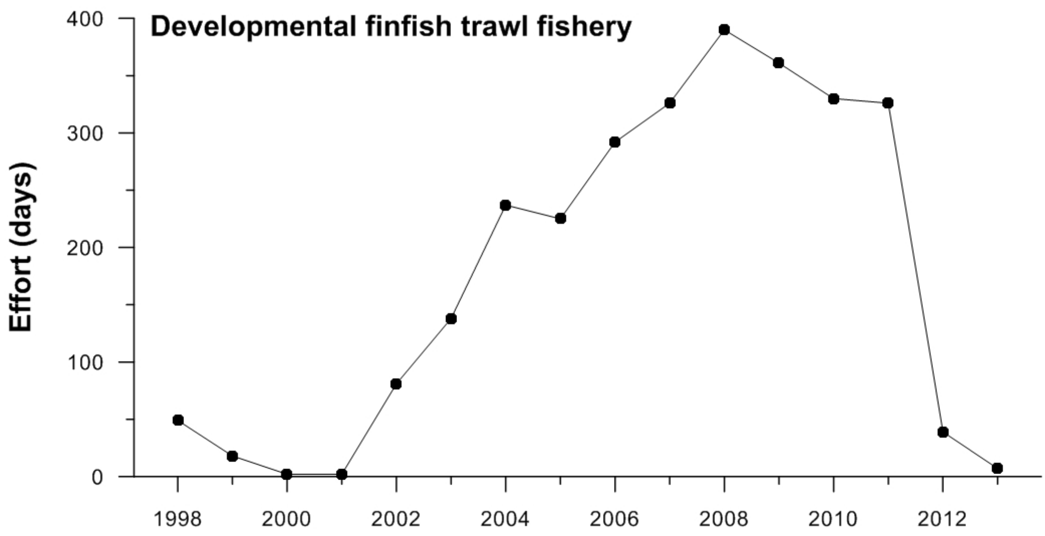

Figure 2.4. Annual fishing effort in the Developmental Fin Fish Trawl Fishery in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

13.3.5 NORTHERN PRAWN FISHERY

The Northern Prawn Fishery (NPF) began in the late 1960s after government and CSIRO surveys indicated the potential for development. Initially targeting White Banana Prawn, *Penaeus merguiensis*, in the south-eastern GoC, this otter trawl fishery expanded over the following decades. Today it landings at least nine species of prawns, as well as byproduct including bugs, squid, scallops and scampi. The fishery now extends over waters from Cape York to the eastern Kimberley coast (Woodhams and Geoge, 2013).

One of Australia's most valuable fisheries, the NPF is essentially two sequential fisheries, one that targets White Banana Prawns, and a mixed species fishery, targeting Tiger (*P. esculentus* and *P. semisulcatus*), Endeavour (*Metapenaeus endeavouri* and *M. ensis*), and Red-legged Banana prawns (*P. indicus*). Banana Prawn annual landings have ranged from less than 2000 t to more than 12,000 t. Tiger Prawn catches peaked in the early 1980s, with annual landings exceeding 5000 t for Grooved Tiger Prawns, and 8000 t in total (Figure 2.5). However, management changes mean that annual catches are much reduced from that peak, with most catches in recent years in the range 1000 to 2300 t. Prawn prices also vary, so the total value of the fishery also changes substantially from year to year. NPF landings in 2011 were valued at $97.1 million but 2012 landings were worth just $64.7 million (Woodhams et al., 2011). In recent years, the White Banana Prawn fishery has provided around two thirds of the Gross Value of Production (GVP) from the fishery (Woodhams et al., 2011) it has been fully fished since the mid-1970s (Lucas et al., 1979; Zhou et al., 2007).

An apparent association between environmental factors, particularly rainfall, and Banana Prawn catches was borne out by statistical analyses (Vance et al., 1985). The strength of the relationships between catches and environmental variables, however, was not consistent over time and varied markedly between regions of the fishery (Vance et al., 1985; 2003). Environmental factors other than rainfall had variable influences on White Banana Prawn catches across regions and between years. Confounding between spatial ecological and operational factors also appeared to generate strong variation between regions in the assessments (Vance et al., 2003).

The White Banana Prawn is highly aggregative, to the extent that large schools stir up sediment. The resulting 'mud boils' can be seen from boats, and early in the fishing season spotter planes are used to target them. Total catches of this species are highly variable from year to year and have varied from less than 2000 t to more than 12,000 t. This variation is related to river flow and rainfall. There has been considerable research into the relationship between catch and flow or rainfall. Unlike Tiger Prawns, which are largely marine, White Banana Prawn juveniles are estuarine. The ecology that underlies the relationship between White Banana Prawn catches and river flow has also been extensively investigated (see Section 8.3).

The management of the fishery is highly sophisticated. The number of vessels in the fishery, as well as the gear that is permitted, is limited, and additional measures such as seasonal and permanent closures provide additional support for management for sustainability and economic performance. The Tiger Prawn fishery is managed using a bioeconomic model (Punt et al., 2011), which not only assesses the status of tiger and endeavour prawn stocks, but provides annual effort targets to ensure that the fishery operates near Maximum Economic Yield (MEY). The White Banana Prawn fishery uses a dynamic catch rate trigger to ensure fishing is near MEY. Successful cooperative management between government and industry has supported Marine Stewardship Council accreditation of the fishery.

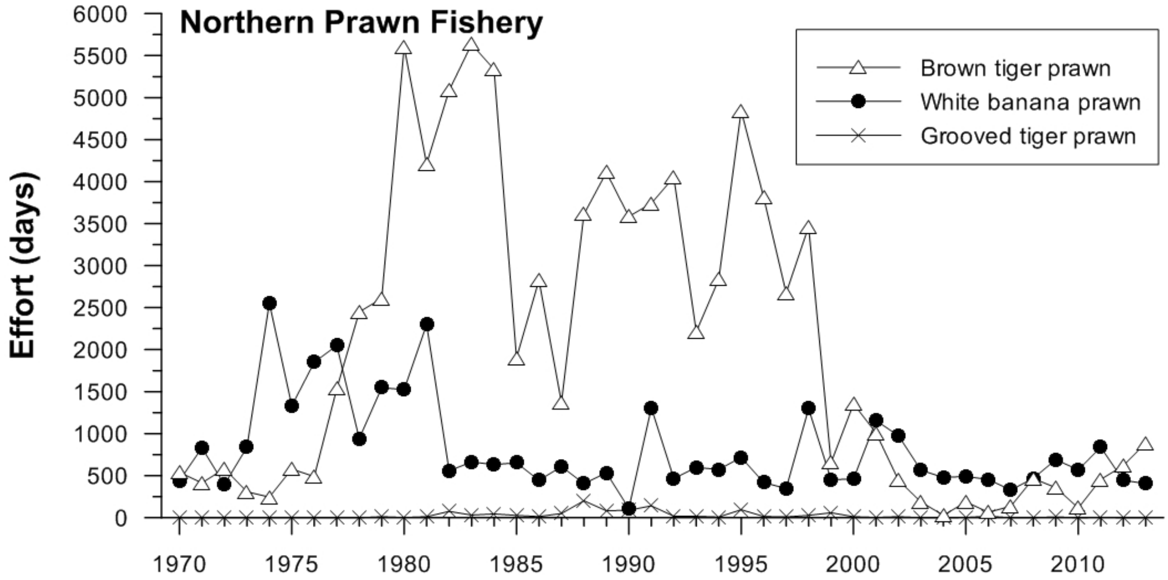

Figure 2.5. Annual fishing effort for Brown tiger, White banana, and Grooved Tiger Prawns in Zone 9 of the Northern Prawn Fishery in the Gulf of Carpentaria. Data supplied with permission by CSIRO Marine and Atmospheric Research.

13.3.6 RECREATIONAL FISHERY

Recreational fishing is a popular sporting and social activity throughout Queensland. Approximately 703,000 Queensland residents participate in the activity and contributed to its total estimated value of $73 m in 2009–2010 (D.E.E.D.I., 2009).

In the study region, there is a diverse range of opportunities for recreational fishers to fish in marine, estuarine and freshwater habitats using a variety of gear types including line, pot, dive and cast net, using either shore-based and boat-based fishing modes. As a result, the study region attracts a high number of recreational fishers, both residents and interstate visitors (Greiner et al., 2013). Several recreational fishing surveys have been undertaken throughout Queensland, but they vary in spatial and temporal extent, methodology, and targeted purpose. In this project, we decided to use data from the 2010 Queensland statewide survey data. It is one of the few studies that collected data over a long period (12 months) using a consistent methodology across our large spatial region of interest and is therefore, the most representative dataset of recreational fishing in the region.

The 2010 survey estimated that around 77,000 fisher days were expended in the study region, comprising the reporting strata of Karumba coastal waters, Mornington Island coastal waters, and the Gulf catchment (i.e. reporting zones 10, 11 and 12). During this survey, it was estimated that fishers in the region caught around 430,000 fish and crustaceans comprising at least 53 species/species groups, which were taken from marine, estuary, freshwater and lake/dam habitats. It is important to note that these catch and effort statistics are for Queensland residents only, and do not take into account the significant seasonal recreational fishing effort that is contributed by interstate fishers (see Greiner et al., 2013), but is not captured in the survey.

Given the diversity of fishing opportunities in the region, there can be large differences in the level and types of fishing activity between river catchments and the adjacent coastal habitats. For example, Taylor et al. (2012) showed that over 95% of fishing activity in Karumba coastal waters was boat-based, whereas nearly 60% of fishing was boat-based in the nearby Gulf catchment. Such differences in fishing activity are likely to reflect differences in habitats and abundance of target species, which cannot be easily teased apart given the coarse spatial resolution of the 2010 statewide survey. These data may not provide absolute certainty of the targeting, catch and effort specifically for the Flinders and Gilbert rivers, but they do provide sufficient information to gauge the general level of importance of particular species to the recreational fishery.

13.3.7 INDIGENOUS FISHERY

Aboriginal and Torres Strait Islander people have lived along the Australian coastline for over 40,000 years. Many of the marine and freshwater species are culturally important for Aboriginal people, and subsistence fishing is a traditional and important food source.

Many northern Australian coastal Aboriginal groups continue to practise customary management and education relating to the sea, being passed to emerging generations through stories, dance, song, art and ceremony. Some Aboriginal groups elect families to act as sea managers, while others allow people to act this role in their mothers' country while residing in their fathers' country. This usually means that Aboriginal people will only fish and hunt within their own country and would seek permission before fishing someone else's country. These are only some of the customary management practices that Aboriginal people use to ensure the sustainability of their resources.

Aboriginal fishers use a wide variety of fishing methods, ranging from spears, hand, line, nets and erection of traditional fish traps using stones. They also fish in a wide range of habitats in freshwater, estuarine, marine, intertidal and supratidal zones depending on the time of year in which local species are abundant and able to be captured.

Unfortunately, there has been very little scientific documentation of Indigenous fishing practices and the composition of the catch. The only study that has collected quantitative data from the fishery is the 2000/01 NRIFS (Henry and Lyle, 2003), which collected data from 44 communities across Australia. For Queensland the survey estimated that at least 10,400 Indigenous persons fished in 2000/01 and undertook 138,000 fishing events and caught 721,511 animals, mostly finfish (346,404 fish).

5.3 Data sources

5.3.1 ESTUARINE AND COASTAL PREDATOR SPECIES

The coastal regions and estuaries of the southeastern GoC hosts a wide range of high level predators that exert top-down predation pressure on prey to help maintain the structure of the ecosystem (Okey, 2006; Pascoe et al., 2008; Griffiths et al., 2010). Many of these large predators are not principal target species of the region's commercial, recreational or Indigenous fisheries, but are important 'keystone' species that may help maintain the structure of the ecosystem. As a result, data were used from fishery independent surveys conducted by the CSIRO throughout the GoC's coastal regions and estuaries to determine important species in an ecological context. These surveys employed a range of sampling methods to account for the selectivity of differently sized fish and species including stake, seine and fyke nets, rotenone, and beam trawl, although the primary method was gillnetting (Blaber et al., 1989; 1990; 1994a; 1995; 2010).

The 50 most abundant species recorded in the Norman River and nearshore surveys (in terms of biomass) are shown in Table 2.1. Interestingly, the relatively small schooling barracudas (*Sphyraena putnamae* and *S. jello*) were among the most abundant species captured, as well as Sawfish (*Pristis microdon* and *P. clavata*), Whaler Sharks (*Carcharhinus leucas* and *C. amblyrhynchos*) and Barramundi (*Lates calcarifer*).

Table 2.1. Highest ranked 50 teleost and elasmobranch species by biomass caught by a range of gear types in the Norman River by Salini et al (1998). Catch rates are expressed as $kg^{-1} hr^{-1}$. Species of conservation importance are denoted by an asterisk (*).

Scientific name	Common name	Beam net	Fyke net	Gill net	Rotenone	Seine net	Stake net	Total
Sphyraena putnamae	Sawtooth Barracuda			7.2000				7.2000
*Pristis microdon**	Freshwater Sawfish			6.5630				6.5620
Carcharhinus leucas	Bull Shark			4.8750				4.8752
Carcharhinus amblyrhynchos	Grey Reef Shark			4.1630				4.1622
Sphyraena jello	Pickhandle Barracuda			3.7000				3.7000
Lates calcarifer	Barramundi		0.6800	1.2741		0.4855	0.4550	2.8946
*Pristis clavata**	Dwarf Sawfish			1.5875				1.5875
Sphyraena qenie	Blackfin Barracuda			1.2500				1.2500
Cynoglossidae spp.	Tongue Soles						1.1250	1.1250
Polydactylus macrochir	King Threadfin	0.0002	0.0169	0.8527			0.2500	1.1197
Hemiarius insidiator	Flat Catfish			0.6025			0.4950	1.0975
Scomberoides commersonnianus	Talang Queenfish			1.0299		0.0084		1.0383
Liza subviridis	Greenback Mullet		0.0149	0.2277	0.0005	0.7253	0.0445	1.0130
Liza tade	Rock Mullet		0.0015	0.6715		0.2711	0.0577	1.0018
Epinephelus tauvina	Greasy Rockcod			0.9762				0.9762
Sciaenidae spp.	Jewfishes		0.0166	0.3629			0.5424	0.9219
Euristhmus nudiceps	Nakedhead Catfish		0.0594	0.7325			0.1290	0.9208
Rhizoprionodon taylori	Australian Sharpnose Shark			0.8753				0.8753
Elops machnata	Australian Giant Herring			0.2034		0.6180		0.8214
Chaetodon trifascialis	Chevron Butterflyfish			0.8125				0.8125
Strongylura urvillii	Urville's Longtom					0.8100		0.8100
Parastromateus niger	Black Pomfret			0.8023				0.8023
Acanthopagrus berda	Pikey Bream			0.3166		0.0752	0.37000	0.7619
Platycephalus indicus	Bartail Flathead		0.0018	0.7500				0.7518
Plotosidae spp.	Eeltail Catfishes			0.7500				0.7500
Eleutheronema tetradactylum	Blue Threadfin		0.0002	0.6957	0.0004	0.0274	0.0230	0.7467
Plicofollis argyropleuron	Longsnout Catfish			0.7095				0.7095
Ariidae spp.	Forktail Catfishes			0.2575		0.0176	0.4150	0.6902
Neoarius graeffei	Blue Catfish		0.0488	0.5046	0.0499	0.0350		0.6383
Platycephalus westraliae	Yellowtail Flathead			0.6250				0.6250
Neoarius leptaspis	Boofhead Catfish		0.0949	0.1802	0.0988	0.179	0.0690	0.6219
Strongylura krefftii	Freshwater Longtom			0.4783		0.0720		0.5503
Mugilidae spp.	Mullets		0.0002	0.4846	0.0418			0.5266
Liza argentea	Goldspot Mullet		0.3550	0.1543		0.0115		0.5209
Leptobrama muelleri	Beach Salmon			0.3509		0.1314	0.0290	0.5113

Scientific name	Common name	Beam net	Fyke net	Gill net	Rotenone	Seine net	Stake net	Total
Escualosa thoracata	White Sardine		0.0019			0.0182	0.4900	0.5101
Paramugil georgii	Fantail Mullet					0.5100		0.5100
Nematalosa erebi	Bony Bream			0.2117		0.2966		0.5083
Cinetodus froggatti	Smallmouth Catfish			0.4875				0.4875
Scomberoides lysan	Lesser Queenfish			0.4520				0.4520
Rhinomugil nasutus	Popeye Mullet			0.1513		0.0132	0.2762	0.4408
Himantura astra	Blackspotted Whipray					0.2237	0.1950	0.4187
Eusphyra blochii	Winghead Shark			0.3700				0.3700
Chelonodon patoca	Milkspot Toadfish		0.0710	0.1113		0.0909	0.0956	0.3688
Chirocentrus dorab	Dorab Wolf Herring			0.3637				0.3637
Otolithes ruber	Silver Teraglin			0.3400				0.3400
Liza melinoptera	Otomebora Mullet			0.3320				0.3320
Marilyna darwinii	Darwin Toadfish		0.0165	0.2733	0.0322			0.3220
Lutjanus johnii	Golden Snapper	0.0190		0.2560		0.0450		0.3200
Epinephelus maculatus	Highfin Grouper			0.3100				0.3100

5.3.2 FORAGE SPECIES

Small mobile fauna that make up the prey of predators in estuaries and coastal habitats can be difficult to sample quantitatively. It is particularly difficult during times of flood as areas can be unsafe or too difficult for deploying sampling gear. Several studies have used stomach content analysis of predators as a means to describe forage fauna that are otherwise difficult to sample (Lansdell and Young, 2007; Potier et al., 2007; Romeo et al., 2012).

The diet of high level predators of GoC estuaries and coastal regions has been studied extensively, primarily in an attempt to quantify the consumption of commercially important penaeids (Brewer et al., 1989; Salini et al., 1990; Brewer et al., 1991; Salini et al., 1992; Brewer et al., 1995; Salini et al., 1998). Therefore, we used the stomach content data of predators caught from the Norman River – a river system adjacent to the Flinders and Gilbert rivers – by Salini et al. (1998) to characterise potentially important forage species that may be affected by altered river flows.

The most abundant prey taxa species recorded in the diets of high level predators in the Norman River and nearshore region (in terms of percentage biomass) are shown in Table 2.2. The most important prey taxa were teleost remains, Mullets, juvenile King Threadfin, White Banana Prawn, catfish, goatfish and Gizzard Shad.

Table 2.2. Forage species constituting more than 1% of the total diet by weight for finfish and elasmobrach predator species caught by a range of gear types in the Norman River by Salini et al (1998).

Scientific name	Common name	% Weight
Teleost remains	Unidentified Teleosts remains	27.865
Mugilidae spp.	Mullets	6.430
Polydactylus macrochir	King Threadfin	5.248
Penaeus merguiensis	White Banana Prawn	5.192
Ariidae spp.	Forktail Catfishes	5.175
Scomberoides commersonnianus	Talang Queenfish	4.610
Mullidae spp.	Goatfishes	4.595
Thryssa spp.	Thryssa Anchovy	4.413
Polychaeta spp.	Polychaete Worms	3.889
Sergestidae spp.	Paste Prawns	2.686
Metapenaeus spp.	Endeavour Prawns	1.765
Pristigasteridae spp.	Ilisha	1.649
Hemiramphidae spp.	Garfishes	1.352
Thryssa hamiltonii	Hamilton's Thyearssa	1.349
Caridea spp.	Shimps	0.867
Bivalvia spp.	Bivalves	0.787
Penaeidae spp.	Penaeid Prawns	0.704
Alpheidae spp.	Pistol Prawns	0.679
Nematalosa come	Hairback Herring	0.669
Mysidae spp.	Mysids	0.595
Metapenaeus eboracensis	York Prawn	0.594
Metapenaeus insolitus	Greasyback Prawn	0.587
Nematalosa erebi	Bony Bream	0.562

5.3.3 KEYSTONE SPECIES

By analysing the previously described datasets, obvious species of importance to the ecosystem could be identified simply by selecting species of highest abundance or biomass. However, in many ecosystems some of the most important ecological interactions occur when species of relatively low biomass or productivity has a disproportionate impact in structuring an ecosystem. These species are known as 'keystone' species (Paine, 1995) and can only be identified by detailed analyses of the trophic relationships between species, or at least ecological 'functional groups' of similar species.

Although extensive dietary studies have been undertaken on the estuarine and nearshore fishes of the GoC, particularly the Norman River (Brewer et al., 1989; Salini et al., 1990; Brewer et al., 1991; Salini et al., 1992; Brewer et al., 1995; Salini et al., 1998), no ecosystem model has been built that explicitly characterises these systems. Building an ecosystem model to identify keystone species is very labour and data intensive, and was therefore beyond the scope of the present project. However, Griffiths et al. (2010) built a trophic mass balance Ecopath ecosystem model of the entire GoC to examine the ecological effects of trawling by the NPF. The core of this model was the diet of nearshore fish, which would therefore allow us to potentially identify nearshore keystone species. The "keystoneness" routine was undertaken in Ecopath software, which ranks each species, or functional group, with respect to its relative impact on the number of other species (and the magnitude) when the species biomass is changed by a small percentage. The keystone index when plotted against the relative total impact (Figure 2.6) allows for the most influential species to be identified.

Figure 2.6 shows that the Tiger Prawn (adult) was the most important keystone species in the GoC Ecopath model. This was due to their high production/biomass ratio (i.e. a measure of productivity) and consumption/biomass ratio (i.e. feeding ration) that is imposed upon a diverse diet of 16 functional species groups within the model. Just as importantly, this species comprises a large part of the diet of many functional groups made up of fish and shark species that occupy a range of trophic levels.

The next most important keystone species grouping were Large Sharks. This functional group was an aggregation of 13 carcharhinid and sphyrinid sharks, but the nominated flagship species were *Carcharhinus leucas, Glyphis glyphis,* and *Eusphyra blochii.* Although each of these species are commonly found in estuaries, it is the former two species that are particularly noteworthy. *Carcharhinus leucas* (Bull Shark) is one of the few catadromous species worldwide that spends a significant proportion of its life in estuaries and often in freshwater (Last and Stevens, 2009). Similarly, *G. glyphis*, the Speartooth Shark, spends a large proportion of its life in freshwater and is of particular conservation importance, currently listed under the EPBC Act.

Benthopelagic carnivores (fish) were the next most important group which was comprised of a range of abundant and highly productive predatory fishes that are common in estuaries and coastal habitats. The group was primarily characterised by the Blue spot Trevally, *Caranx bucculentus*, which has been documented to consume a high biomass of penaieds and other small fish in the GoC (Brewer et al., 1989).

Benthic invertebrate feeding fish were also very important in influencing the community structure. These are generally small schooling fish that extend from the estuaries to offshore regions with the functional group being characterised by families including grunter, goatfish, trumpeter, Jewfish and Threadfin. The primary species in this group is the Largescale grunter, *Terapon theraps.*

Benthopelgic invertebrate-feeding fish contributed 80% of the total relative impact on community structure. This group is also comprised of small schooling fish species that generally occupy regions higher in the water column, in particular Hamilton's Thyearssa, *Thryssa hamiltonii*.

Interestingly, small gastropods and zooplankton also ranked highly as keystone groups. The high ranking of zooplankton may indicate that the ecosystem is driven by bottom-up processes as the relatively small biomass of this low trophic level group plays a large role in supporting a range of predators higher in the food web.

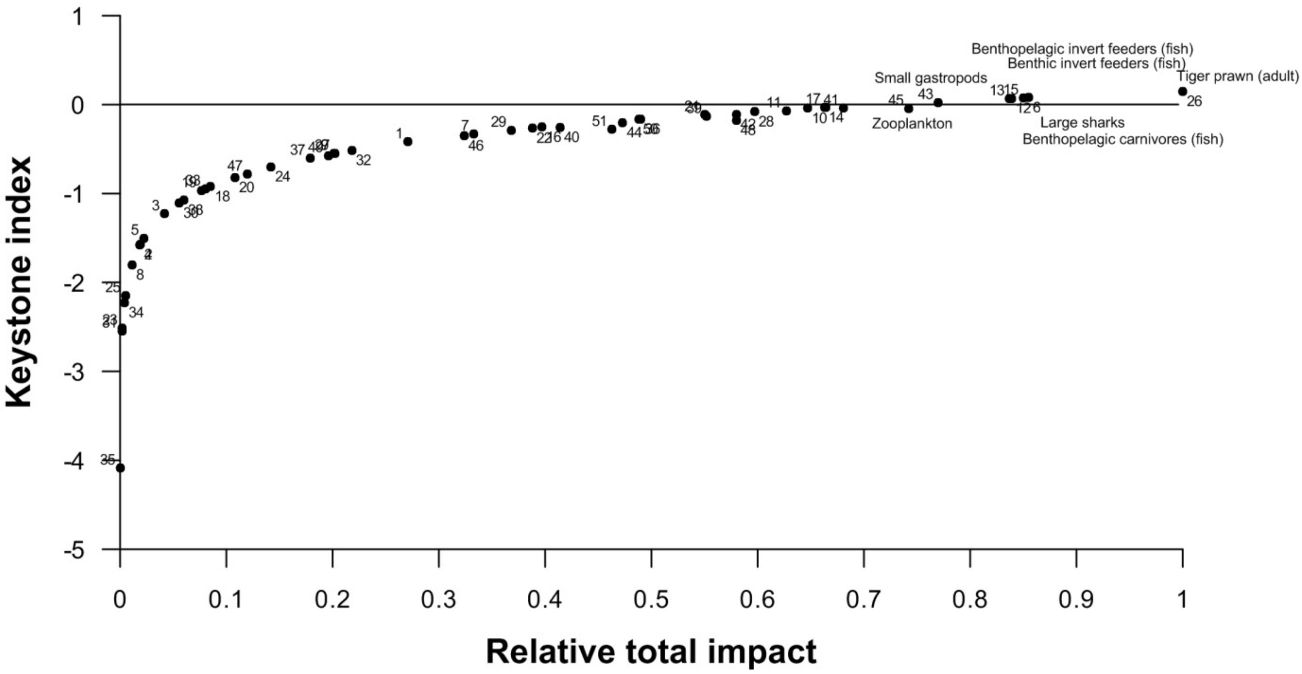

Figure 2.6. Identification of keystone species using the keystone index of each ecological functional group plotted against their relative total impact on the biomass of other functional groups as determined within an Ecopath ecosystem model of the Gulf of Carpentaria (Griffiths et al. 2010). The most important functional groups for structuring the ecosystem are shown to the right of the graph accompanied by a label.

5.3.4 COMMERCIAL FISHERIES

The coastal regions and estuaries of the south-eastern GoC host a wide range of high-level predators that exert top-down predation pressure on prey to help maintain the structure of the ecosystem (Okey, 2006; Pascoe et al., 2008; Griffiths et al., 2010). Some of these large predators also support some of the region's most important commercial, recreational and Indigenous fisheries, and therefore some research on the movement and dynamics of their populations has been undertaken to support management (Welch et al., 2009), stock assessments (Dichmont et al., 2003; Griffiths et al., 2006) or to monitor catch levels (Kenyon et al., 2011).

The data from commercial logbooks (Table 2.3) and scientific observer surveys (Table 2.4) were used to select the highest priority species for inclusion in the qualitative risk assessment. Species were first chosen with regards to importance by catch weight or numbers, but this list was complemented with species of known 'iconic' importance, which may not be captured in large numbers, but have significant social and/or economic value.

5.3.5 RECREATIONAL FISHERIES

Of the 50 species most commonly caught by recreational fishers in the study region (Table 2.5), the iconic Barramundi was caught in the highest numbers (125,000 fish), and is generally caught in estuaries and to a lesser extent in freshwater. Pikey Bream (48,539) was the next most commonly caught species, followed by Barred javelin (i.e. 'Grunter') (37,657), Forktail catfish, Blue and King Threadfin and Mud Crab. The most commonly caught species that are generally found in freshwater were Sooty Grunter, Eeltail catfish, Archerfish and Spangled perch.

Based on catch data and iconic status (Table 2.5), the species selected for further investigation in qualitative risk assessments were: Barramundi, Pikey Bream, Grunter, Sooty Grunter, Blue and King Threadfin, Mud Crab and Mangrove Jack.

5.3.6 INDIGENOUS FISHERIES

Fishing is a highly important social and cultural activity for the Aboriginal people around the GoC. It is also provides a major food source for local Aboriginal communities, and thus Indigenous fishing should be regarded as subsistence fishing, rather than fishing for recreation. Unfortunately, there has been very little scientific documentation of the composition and relative abundance of Indigenous catch across northern Australia.

There have been some attempts at eliciting knowledge from some Aboriginal communities with respect to the ecology and movements of common species and their importance to culture and fishing (Close et al., 2014; Jackson et al., 2014). However, these studies do not provide detailed species-specific information required in this study to prioritise species for inclusion in qualitative risk assessment.

The only study that has collected such data was the 2000/01 NRIFS (Henry and Lyle, 2003). Although data from this study were available for Queensland only, it provides reasonably detailed data that can, with the help of expert opinion, be used to prioritise important species for the Indigenous fishery. The total Indigenous catch by taxonomic group from the NRIFS is shown in Table 2.6.

5.3.7 FINAL LIST OF PRIORITY SPECIES

Following the analysis of fishery catch data, fishery independent survey data, and the results from a quantitative ecosystem model of the GoC, a total of 46 species (or species groups) were identified for inclusion in formal qualitative risk assessments. Table 2.7 shows the relative importance of each species to particular fisheries, the ecosystem, and the conservation values. The recreational fishery had the highest number of species to be assessed, primarily due to the range of habitats fished and fishing methods used in the GoC. The N3 fishery was the commercial fishery with the highest number of species to be assessed, primarily due to the capture of a range of commercially important offshore, coastal and estuarine species.

Table 2.3. Mean annual catch (in tonnes) for 2009–2013 for the 50 highest ranked species (by weight) recorded in commercial fishery logbooks for all Queensland fisheries combined in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

Common name	Mean annual catch (t)
Barramundi	792.97
Grey Mackerel	697.85
King Threadfin	300.22
Spanish Mackerel	239.91
Mud Crab	183.86
Crimson Snapper	181.60
Saddletail Snapper	135.00
Australian Black-tip Shark	107.44
Black-tip whalers	80.02
Blue Threadfin	52.48
Jewel	32.88
Mangrove Jack	25.00
Sorrah Shark	19.75
Goldband Snapper	19.20
Hammerhead Shark	18.16
Unspecified Shark	15.54
Golden Snapper	15.20
Unspecified Queenfish	15.19
Painted sweetlip	14.40
Unspecified grunter	14.03
Unspecified Threadfin	11.04
Blue Catfish	10.80
Red Emperor	6.00
Unspecified Jewfish	5.85
Black kingfish	5.22
Winghead Shark	3.62
Unspecified pomfret	3.28
Bull Shark	3.14
Baitfish	3.01
Unspecified whaler Shark	2.69
Unspecified tuna	2.67
Unspecified Trevally	1.93
Flat Threadfin	1.64
Shovelnose guitarfishes	1.51
Unspecified catfish	1.47
Striped seapike	1.42
Moses perch	1.38
Unspecified Mullet	1.37
Blue Swimmer Crab	1.24
Red Emperor	1.18
Milkfish	1.08
Creek whaler	1.01
Unspecified garfish	0.94
Golden catfish	0.91
Black-tip whaler Shark	0.79
Unspecified tropical Snapper	0.76
Milk Shark	0.42
Spinner Shark	0.39
Tiger Shark	0.32
Unspecified crab	0.32

Table 2.4. Species composition and number of fish recorded by scientific observers monitoring the catches of Queensland gillnet fisheries (N3, N12 and N13) for 2009–2012 in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

Common name	Number
King Threadfin	688
Barramundi	374
Black Pomfret	256
Winghead Shark	223
Blue Threadfin	208
Grey Mackerel	198
Giant Queenfish	136
Whitecheek Shark	92
Black-tip Whaler Complex	72
Milk Shark	71
Bull Shark	71
Creek Whaler	47
Striped Scat	46
Scaly Jewfish	34
Spot-tail Shark	32
Mud Crab	27
Hawaiian Giant Herring	23
Forktail Catfishes	20
Trevallies	16
Dorab Wolf Herring	14
Spotted Scat	14
Gizzard Shad	13
Beach Threadfin	13
Silver Jewfish	11
Barred Javelin	10
Spanish Mackerel	9
Scalloped Hammerhead	8
Giant Shovelnose Ray	8
Australian Halibut	7
Tuna and Mackerel	6
Great Hammerhead	5
Tripletail	5
Speartooth Shark	4
Guitarfishes and Wedgefishes	4
Milkfish	4
Cobia	4
Barred Queenfish	4
Black Jewfish	4
Balmain Bugs	3
Lemon Shark	3
Banded Eagle Ray	3
Whitespotted Eagle Ray	3

Table 2.5. Annual estimated catch numbers (includes retained and released fish) of the species/species groups caught by recreational fishers in the study region for 2010. Only species with catch estimates considered to be of "Medium" or higher in terms of estimation reliability have been included. Data derived from Taylor et al. (2012) for recreational fishers completing a 12-month fishing diary for trips undertaken in reporting regions 10, 11 and 12 and then scaled to the population level to provide catch estimates herein. Species considered to be iconic in the recreational fishery in the region are denoted by an asterisk.

Common name	Catch	Confidence
Barramundi	125000	Medium
Pikey Bream	49000	Medium
Barred javelin*	38000	Medium
Forktail catfish - unspecified	21000	Medium
Sooty Grunter*	19000	Medium
Blue Threadfin*	17000	Medium
Mud Crab*	13000	Medium
Eeltail catfish - unspecified	12000	Medium
Jewfish - unspecified	12000	Medium
Yellowfin bream	10000	Medium
King Threadfin*	7000	Medium
Cod and groper - unspecified	5000	Medium
Mangrove Jack*	4000	Medium
Queenfish	4000	Medium
Archerfish	2000	Medium
Coral trout - unspecified	2000	Medium
Golden Snapper	1000	Medium
Spangled perch	1000	Medium
Spanish Mackerel*	1000	Medium

Table 2.6. Annual harvest (in numbers) and percentage contribution of species/species groups by Indigenous fishers, aged five years or older, in Queensland waters during 2001. Data reproduced from Henry and Lyle (2003).

Common name	Number caught	Contribution (%)
Prawns (saltwater)	131,158	18.18
Pipi/ Goolwa cockle	71,607	9.92
Small baitfish	71,012	9.84
Mullet	68,573	9.50
Bream	44,205	6.13
Sea perch/Snappers	38,200	5.29
Oysters	34,615	4.80
Garfish	26,169	3.63
Catfish	21,738	3.01
Trevally	21,494	2.98
Whiting	19,879	2.76
Other finfish	19,321	2.68
Grunters/trumpeters	15,116	2.10
Other shellfish	13,124	1.82
Lobsters	12,903	1.79
Mud Crab	12,874	1.78
Threadfin Threadfin	11,950	1.66
Cod (various)	11,679	1.62
Emperors	9,268	1.28
Wrasse/tuskfish/gropers	9,181	1.27
Coral trout	7,004	0.97
Barramundi	5,745	0.80
Turtle eggs	3,976	0.55
Turtle - saltwater unspec.	3,851	0.53
Sharks/rays	3,819	0.53
Herring/pilchards	3,545	0.49
Mussels	3,499	0.48
Turtle - freshwater unspec.	3,243	0.45
Flathead	2,384	0.33
Mackerels	2,382	0.33
Crabs (other)	2,345	0.33
Crayfish (freshwater)	2,276	0.32
Butterfish	2,189	0.30
Non-fish - other	1,690	0.23
Dugong	1,293	0.18
Turtle - longneck	1,214	0.17
Red Emperor	1,207	0.17
Pike	972	0.13
Blue Swimmer Crab	882	0.12
Eels	869	0.12
Redfish	795	0.11
Pink Snapper	726	0.10
Australian bass/perch	612	0.08
Mulloway/Jewfish	366	0.05
Dart	207	0.03
Leatherjackets	176	0.02
Tailor	97	0.01
Luderick	80	0.01

Table 2.7. Identified priority species for inclusion in qualitative risk assessments to determine the potential impact of altered flow regimes on their long term sustainability. Shading of cells indicates the relative importance of each species to particular values. Dark grey = High importance (e.g. principal target or iconic species); Mid grey = important (e.g. byproduct species); Light grey = low importance (e.g. occasional byproduct species); White = not important.

	Commercial						Recreational	Indigenous	Listed	Ecosystem
Species	NPF	Offshore net	Inshore net	Line	Fish Trawl	Mud Crab				
Barramundi										
Pikey Bream										
Grunter										
Sooty Grunter										
Blue Threadfin										
King Threadfin										
Mullet										
Mud Crab										
Blue Swimmer Crab										
Mangrove Jack										
Crimson Snapper										
Saddletail Snapper										
Red Emperor										
Golden Snapper										
Goldband Snapper										
White Banana Prawn										
Grooved Tiger Prawn										
Brown Tiger Prawn										
Slipper Lobsters										
Cephalopods										
Grey Mackerel										
Spanish Mackerel										
Black-tip Shark										
Black Jewfish										
Blue-spot Trevally										
Talang Queenfish										
Blue Catfish										
Freshwater Sawfish										
Sea snakes										
Green Sawfish										
Narrow Sawfish										
Speartooth Shark										
Northern River Shark										
Freshwater Whipray										

	Commercial						Recreational	Indigenous	Listed	Ecosystem
Species	**NPF**	**Offshore net**	**Inshore net**	**Line**	**Fish Trawl**	**Mud Crab**				
Largescale Terapon										
Sawtooth Barracuda										
Bony Bream										
Hamilton's Thryssa										
Bull Shark										
Winghead Shark										
Hammerhead Sharks										
Saltwater crocodile										
Dugong										
Marine Turtles										
Migratory Shorebirds										
Plankton										

6 Qualitative Risk Assessment for Species, Habitats and the Ecosystem

6.1 Background

Assessing ecological impacts by natural or anthropogenic perturbations in tropical ecosystems is difficult. Not only is it difficult owing to the high diversity of species and life histories that are generally present in tropical regions of the world (Blaber, 2000; Stobutzki et al., 2001b; Blaber, 2002) and the cost of quantitatively studying each population, but also due to the complexity and computational difficulties of modelling an ecosystem (Pauly et al., 2000). As a result, a range of qualitative to quantitative methods have been developed to make use of available data to assess the risk that particular perturbations will manifest into negative impacts on ecological, social or economic values. The process, called risk assessment, can be defined as the process of estimating the probability – either in qualitative or quantitative terms – that an adverse event with specific consequences will occur in a given period (Lindenmayer and Burgman, 2005).

Risk assessment has been increasingly employed in natural resource management worldwide (Suter, 1993). Ecological risk assessment is a logical process for objectively defining the risk(s) and the probability of an adverse effect upon an organism, or collection of organisms, when exposed to one or more environmental or anthropogenic stressors (Newman, 2001). Ecological risk assessment applications range from biosecurity (Pheloung et al., 1999) and ecotoxicology on populations of single species and ecosystems (Newman, 2001; Pastorok et al., 2002) to conservation biology and fisheries (Burgman et al., 1993; Punt and Walker, 1998; Cheung et al., 2005).

6.1.1 RECENT APPROACHES TO RISK ASSESSMENT IN FISHERIES

Risk assessment in fisheries has been a particularly problematic area given the difficulty and expense in collecting long-term data that can be used to forecast the potential effects of perturbations. Consequently, a number of approaches have been developed in recent years to allow ecological risk assessments of data-limited fisheries.

Qualitative approaches

Fletcher (2005) and Hobday et al. (2006) developed similar qualitative likelihood–consequence approaches modified from the Australian and New Zealand Standard Risk Analysis (Standards Australia, 2000) (Standards, 2000). They assessed the risk of a broad range of fishing-related activities to the sustainability of specific components of fisheries supporting ecosystems, such as target species, byproduct, bycatch and general ecosystem integrity and functionality. The approach is largely facilitated by stakeholder groups agreeing on the likelihood and possible consequence of a particular risk occurring in the fishery.

The advantage of qualitative approaches is that they are rapid, cost-effective and require little data. Because the method relies heavily on stakeholder involvement, management strategies arising from assessments are more likely to be accepted by stakeholders. These methods are useful in scoping assets at potential risk from an activity, but they cannot directly assess whether a species or assemblage will be sustainable following the assessed perturbation.

Semi-quantitative models

A number of semi-quantitative attribute-based ecological risk assessment methods have been developed to assess the relative sustainability of individual species impacted by fisheries. These include Susceptibility-Recovery Analysis (SRA) (Milton, 2001; Stobutzki et al., 2001a), Fuzzy Logic Expert Systems (Cheung et al., 2005), Productivity-Sustainability Analysis (PSA) (Hobday et al., 2011) and Qualitative Risk Matrices (Astles et al., 2006). These methods are similar in that they rank each species on a number of criteria relating to their susceptibility to being captured, and their capacity to recover should the population become depleted. For each species, susceptibility criteria (e.g. geographic distribution, water column position and diel migration) and recovery criteria (e.g. reproduction strategy, growth rate and fecundity) are given a rank reflecting the contribution of the attribute to the overall sustainability of the species. The species having the lowest ranks across all criteria are then considered the highest risk species.

These methods have the advantage of being able to assess the relative sustainability of hundreds of species with little quantitative biological or catch data. As a result, such methods have been popular for assessing the sustainability of diverse, rare and low value species impacted by GoC trawl and gillnet fisheries (Stobutzki et al., 2001a; Gribble et al., 2004). Unfortunately, these semi-quantitative attribute based methods require a great deal of biological information for each of the hundreds of species that are likely to be in some way affected by altered river flows in northern rivers. Such information is available for some species, but such a task is simply not possible given the time and resources available to this project.

More recently, Zhou and Griffiths (2008) developed a more powerful ecological risk assessment method for fisheries that: i) optimises the use of the limited data available for the vast majority of non-commercial species, ii) is conceptually simple, and iii) delivers a quantitative assessment of impact, which provides greater confidence that high risk species are in fact at high risk of depletion. The method, Sustainability Assessment of Fishing Effects (SAFE), is a quantitative modelling approach using similar 'susceptibility' and 'recovery' concepts as existing semi-quantitative attribute-based methods. However, it places greater emphasis on susceptibility elements, particularly spatial distribution of a species, which can be modelled from simple detection–nondetection data recorded in any type of survey (Zhou and Griffiths, 2007). The SAFE method was initially developed for assessing the ecological sustainability of bycatch in the NPF, but has since been applied to several fisheries in Australia and internationally. Because SAFE is not as rapidly applied as other semi-quantitative risk assessment methods, it was not selected to assess the impacts of altered flows. However, it may be tailored to assess flow impacts on individual species rather than fisheries and would be a cost-effective quantitative method to apply in future investigations of the effects of altered river flows on speciose assemblages.

Quantitative ecosystem models

Ecosystem models are one of the few tools that can characterise entire ecosystems and predict the consequences of specific perturbations impacting the system. Unfortunately, these models require intimate knowledge of the entire system and high resolution data relating to the biology of species and their trophic linkages. This usually requires several years of intense field collections followed by long periods to construct and validate the models. Constructing such a model for the Flinders–Gilbert catchment was beyond the resources of the project, but previous CSIRO projects have developed ecosystem models of the GoC to address a range of fisheries-related issues (Okey, 2006;

Okey et al., 2007; Griffiths et al., 2010). The most recent model developed by Griffiths et al. (2010) was designed to assess benthic trawl impacts from the NPF on ecosystem functionality. Although the model characterises the entire GoC, it does allow some exploration of possible flow impacts on specific coastal taxa. Although the model aggregates ecologically similar species ‘functional groups’, the model can be used as a means of identifying potential keystone species in our study area, given the distribution of many species included in the model extend into estuarine habitats.

6.2 The qualitative risk assessment approach

Due to the large number of species potentially affected by reduced river flows, it is not feasible to quantitatively assess hundreds of species given the time and resources available to this project. Therefore, the qualitative likelihood–consequence risk assessment approach developed by Fletcher (2005) as the national fisheries-ESD reporting framework was used. The approach has been adapted from the Australian and New Zealand Standard Risk Analysis and has been applied extensively across Australian fisheries. In particular, this approach has already been used by the Queensland Government to assess GoC fisheries for Wildlife Trade Operation certification under the EPBC Act (Zeller and Snape, 2006).

The method assesses the relative risk of a specific perturbation – in this case fishing-related activities – to the sustainability of specific components of fisheries and the ecosystem, including target, byproduct, and listed species, as well as ecosystem integrity and function. The assessment is primarily driven by expert opinion and stakeholder engagement to assess the potential consequence of a particular risk occurring and the likelihood that this consequence would actually manifest. This approach allows a rapid, flexible and transparent qualitative risk assessment process, with a primary goal of prioritising the species of highest risk of being negatively affected by reduced flows in the Flinders and Gilbert rivers.

The ecological risk assessment was undertaken in three stages:

1. Identify relevant issues by creating a 'Generic ESD Component Tree' and take the branches from this tree to create a subset of component trees, with each specific to a fishery (e.g. NPF), conservation values (e.g. threatened, endangered or protected species), or ecological values (e.g. ecosystem integrity).

2. Objectively prioritise which species to include, then use the risk assessment to determine which species or values are likely to be impacted by reduced flows, to the extent that they will require management or mitigation measures.

3. Complete appropriately detailed reports on each potentially at-risk species and fully justify for finalised risk scores and recommendations.

Identification of potentially at-risk assets

The risk assessment framework of Fletcher (2005) has a number of categories for characterising elements of the broad issues being assessed (e.g. 'contributions to the ecosystem', 'contributions to human wellbeing'). This allows for issues to be disaggregated into manageable components, such as risk to individual species.

To establish a starting point from which to identify the key issues, a generic 'component tree' for the issue of 'Altered River Flows' was constructed (Figure 2.7). The tree has two main components, 'Ecosystem' and 'Fisheries'. Within the 'Ecosystem' branch, the key values were 'Species of Conservation Importance' and 'Ecosystem Integrity'. Within the 'Species of Conservation Importance' branch, two components were defined for species that were listed under the EPBC Act and/or the IUCN Red List. Under the 'Ecosystem Integrity' branch, components were defined largely relating to trophic function as 'Predators', 'Forage', or 'Keystone' species (Figure 2.8). Both 'Species of Conservation Importance' and 'Ecosystem Integrity' components are later used to develop their own component trees, where individual species were identified and formally assessed.

Within the Fisheries branch, the key values that could be compromised by altered river flows are obviously species of economic and social importance across the breadth of fisheries in the region. These are target species, and species that are caught incidentally when attempting to catch the target species and are of economic value, herein termed 'byproduct'. This differs to 'bycatch' species that are also caught incidentally, but discarded. Bycatch is not explicitly listed as a component in this assessment as the objective of the 'Fisheries' branch is to assess the risk of altered river flows on aspects that may directly impact fisheries. Although bycatch species may not be important economically, many play a key ecological role and are identified and assessed under the 'Ecosystem' branch.

The next stage under the 'Fisheries' branch was to broadly identify the fisheries likely to be impacted by altered river flows. The fisheries identified were 'Commercial', 'Recreational' and 'Indigenous' (Figure 2.9). Because the latter two fisheries cannot be further subdivided, they each progress to the final stage of having a component tree where individual species are identified for final assessment. For the 'Commercial fishery' component, six fishery components were identified, the NPF, and the five Queensland fisheries (N3, N12/N13, Mud Crab, Line, Developmental fin fish trawl) (Figure 2.9). An individual component tree was then constructed for each of the six identified fisheries for the final assessment. Within each of these trees subcomponents were constructed for target species and byproduct species in order to clearly identify the species of importance to each fishery that may be affected by altered river flows.

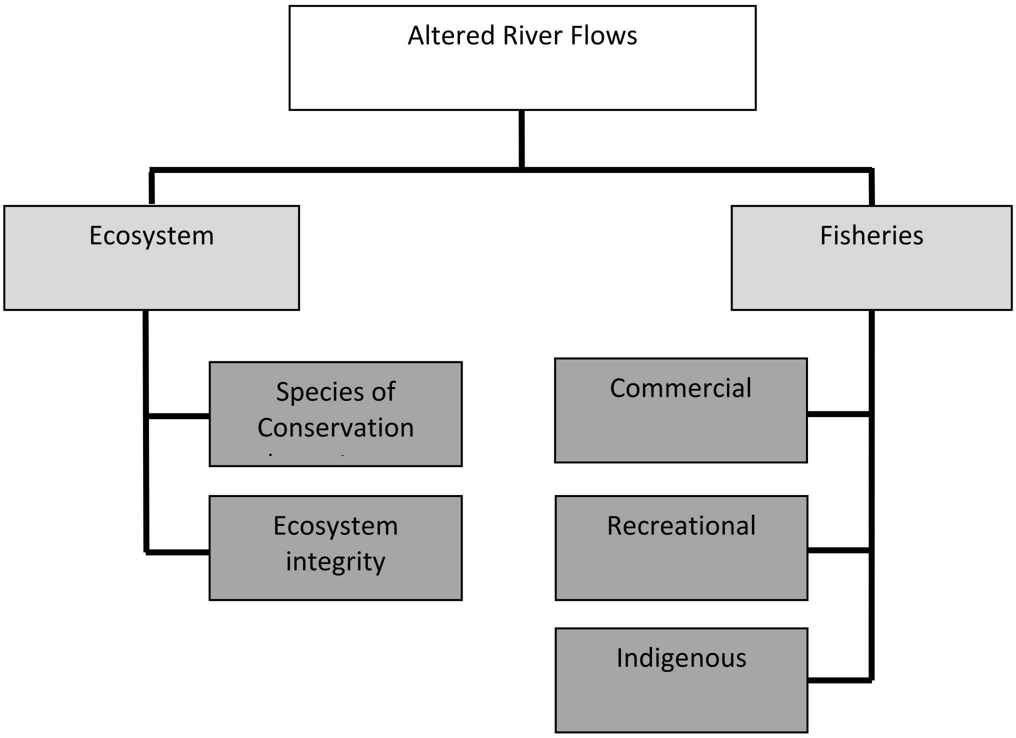

Figure 2.7. The generic component tree identifying the ecosystem and fisheries as the key components for qualitative risk assessment.

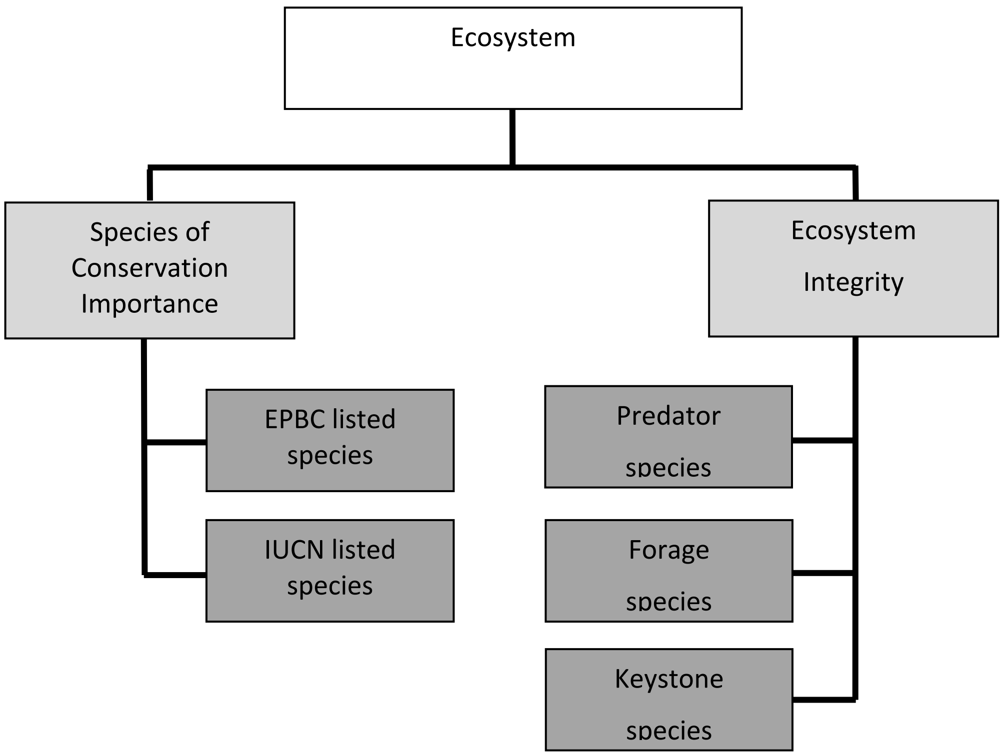

Figure 2.8. The generic component tree identifying the ecosystem components for qualitative risk assessment.

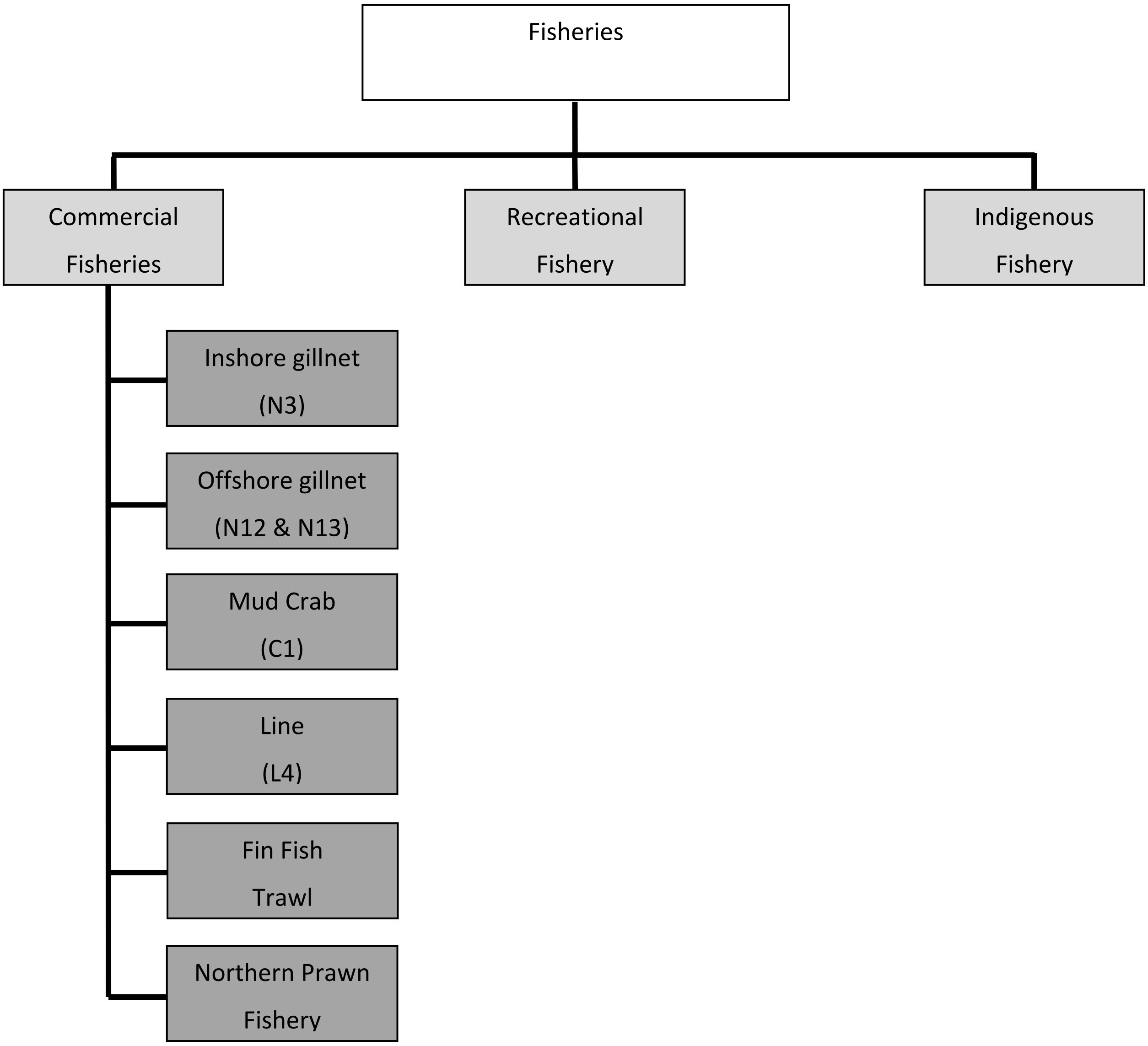

Figure 2.9. The generic component tree identifying the fisheries components for qualitative risk assessment.

Risk assessment of identified priority species

On completion of the 10 component trees, each node within an individual tree needed to be populated with species of importance to that node (e.g. the byproduct node of the N3 fishery component tree). This was done objectively by selecting species based on available data, expert opinion and feedback from stakeholder groups (see "Data sources"). For each of the identified species, a process to prioritise each of these was completed using a formal risk assessment process. The risk assessment framework was consistent with the Australian Standard AS/NZS 4360:1999 for Risk Management. In summary the risk assessment process considers the range of potential consequences of altered river flows on the populations of species and how likely those consequences are to occur. The combination of the level of consequence and the likelihood was used to produce a *relative level of risk* associated with altered river flows. In this case, the specific threat was defined for the most pessimistic water extraction scenario, which was the installation of the Dagworth and Greenhill dams in the Gilbert River, while simultaneously applying the On-farm 560 GL extraction scenario in the Flinders River. This scenario was estimated to reduce the downstream water flow by approximately 19% and 40% of the median water year flow in the Flinders and Gilbert rivers, respectively. This scenario was chosen as a precautionary approach in that species identified as high risk would most likely be at lower risk given a reduction in water extraction levels.

Therefore, the question being proposed in the qualitative risk assessment was:

"***What is the relative level of risk to a species population or habitat value if the level of water extraction was reduced to 19% and 40% of the median water year flow in the Flinders and Gilbert rivers?***"

Consequence criteria for assessment of either species (Table 2.8) or habitats (Table 2.9) were defined and given a value from 0 to 5, with 0 having a negligible consequence, to 5 having catastrophic and irreversible consequence. Because this project primarily deals with potential flow impacts on species of importance to fisheries, many of which extend far beyond the Gilbert and Flinders river delta, it was necessary to define *the scale of potential impact at the species population (or stock) level and not at the level scale of the estuary*. For example, a species that has a widespread distribution consisting of a single stock throughout Australian waters and reproduces outside of the Gilbert or Flinders rivers would be given a far lower consequence score than a species that is endemic to the rivers and completes its entire life cycle within them. A similar approach was used when assessing habitats such as seagrass and mangroves.

The likelihood that a specific consequence would occur was divided into six criteria ranging in score from 1 (remote) to 6 (likely) (Table 2.10).

Expert scientists who have intimate knowledge of the species and habitats assessed determined the consequence and likelihood scores. These scores were presented to stakeholders and other external experts in Workshop 2 and out of session. Scores were only changed in two cases, freshwater sawfish and shorebirds, but this did not change their overall risk ranking.

The overall risk score was calculated by multiplying the consequence and likelihood scores and arriving at one of five risk categories defined by a different colour for ease of interpretation (Table 2.11). The categories are and Negligible, Low, Moderate, High, and Extreme. For each risk category the scores, the recommended management response and the required level of reporting within this project are defined (Table 2.12). Species categorised as Negligible or Low risk do not require the recommendation of a specific management response, but need to be included in the reporting with a short justification as to how the species' life history influenced the risk scores. Moderate and High risk species generally require a recommendation of further analysis and/or further data collection, as well as a detailed justification of risk scores. Species at Extreme risk (of which there were none in this project) require immediate management action (e.g. EPBC listing, spatial/temporal closures) and a detailed report, such as a quantitative population assessment or threat abatement plan. Regardless of its risk score in this project, each species included in the qualitative risk assessment was subject to a literature review as the primary means of justifying the risk score.

After calculating the risk score for each species, the corresponding risk colour was applied to the box for the species in the corresponding component tree(s). This allows each component tree to be displayed where stakeholders can easily see which species were assessed and their relative risk scores.

Table 2.8. Consequence scores and associated criteria for the impact of reduced river flows on the long-term sustainability of species populations.

Score	Criteria
Negligible (0)	Insignificant impacts on the population. Unlikely to be measurable against background variability for the population.
Minor (1)	Possibly detectable, but minimal impact on population size and none on dynamics (e.g. age or genetic structure).
Moderate (2)	Detectable change in population, but long-term recruitment/dynamics not adversely impacted.
Severe (3)	Affecting long-term recruitment levels of stocks/or their capacity to increase.
Major (4)	Likely to cause local extinction, if continued in longer term (i.e. requires listing of species under EPBC or IUCN category).
Catastrophic (5)	Extinction is imminent or immediate

Table 2.9. Consequence scores and associated criteria for the impacts of altered river flows on habitat sustainability.

Score	Criteria
Negligible (0)	Insignificant impacts to habitat or populations of species making up the habitat – probably not measurable levels of impact. Activity only occurs in very small areas of the habitat, or if larger area is used, the impact on the habitats from the activity is unlikely to be measurable against background variability. (Suggestion-these could be activities that affect < 1% of original area of habitat or if operating on a larger area, have virtually no direct impact)
Minor (1)	Measurable impacts on habitat(s) but these are very localised compared to total habitat area. (Suggestion – these impacts could be < 5% of the original area of habitat)
Moderate (2)	There are likely to be more widespread impacts on the habitat but the levels are still considerable acceptable given the % of area affected, the types of impact occurring and the recovery capacity of the habitat. (Suggestion – for impact on non-fragile habitats this may be up to 50% [similar to population dynamics theory] -but for more fragile habitats, to stay in this category the percentage area affected may need to be smaller, e.g. 20%).
Severe (3)	The level of impact on habitats may be larger than can reasonably ensure that the habitat will be able to recover adequately, or it will cause strong downstream effects from loss of function. (For example, where the activity makes a significant impact in the area affected and an area > 25-50% [based on recovery rates] of habitat is being removed).
Major (4)	Substantially too much of the habitat is being affected, which may endanger its long-term survival and result in severe changes to ecosystem function. (For example, this may equate to 70 to 90% of the habitat being affected or removed by the activity.)
Catastrophic (5)	Effectively the entire habitat is in danger of being affected in a major way/removed. (For example, this is likely to be in range of > 90% of the original habitat area being affected.)

Table 2.10. Likelihood scores and criteria for assessing the potential consequences of the impact of reduced river flows on species populations and habitats.

Score	Criteria
Remote (1)	Never heard o,f but not impossible
Rare (2)	May occur in exceptional circumstances
Unlikely (3)	Uncommon, but has been known to occur elsewhere
Possible (4)	Some evidence to suggest this is possible here
Occasional (5)	May occur
Likely (6)	It is expected to occur

Table 2.11. Risk matrix used to obtain overall risk rating for each component based on the consequence of a risk and the likelihood that the consequence will occur.

		Consequence					
		Negligible	Minor	Moderate	Severe	Major	Catastrophic
Likelihood		0	1	2	3	4	5
Remote	1	0	1	2	3	4	5
Rare	2	0	2	4	6	8	10
Unlikely	3	0	3	6	9	12	15
Possible	4	0	4	8	12	16	20
Occasional	5	0	5	10	15	20	25
Likely	6	0	6	12	18	24	30

Table 2.12. Risk ranking outcomes for each component including the response and reporting requirements for each risk category.

Risk	Scores	Recommended response	Reporting
Negligible	0-2	No response	Note inclusion in assessment
Low	2-6	No response	Short justification required
Moderate	5-12	Further literature and/or data analysis required	Full justification required
High	12-24	Further data collection and analysis required	Full justification required
Extreme	20-30	Immediate management required	Detailed performance report

7 Results – qualitative risk assessment component trees

7.1 Northern Prawn Fishery

The important species of the NPF can be divided into target and byproduct species (Figure 2.10). The target species included were the five principal target species in the fishery that are caught in the GoC. Red-legged Banana Prawn was the only species not included as it is only caught in the Joseph Bonaparte Gulf. Two species groups were included as byproduct, Slipper Lobsters (*Thenus parindicus* and *T. australiensis*) and Cephalopods, which includes several Squid (Loliginidae) and Cuttlefish (Sepiidae) species.

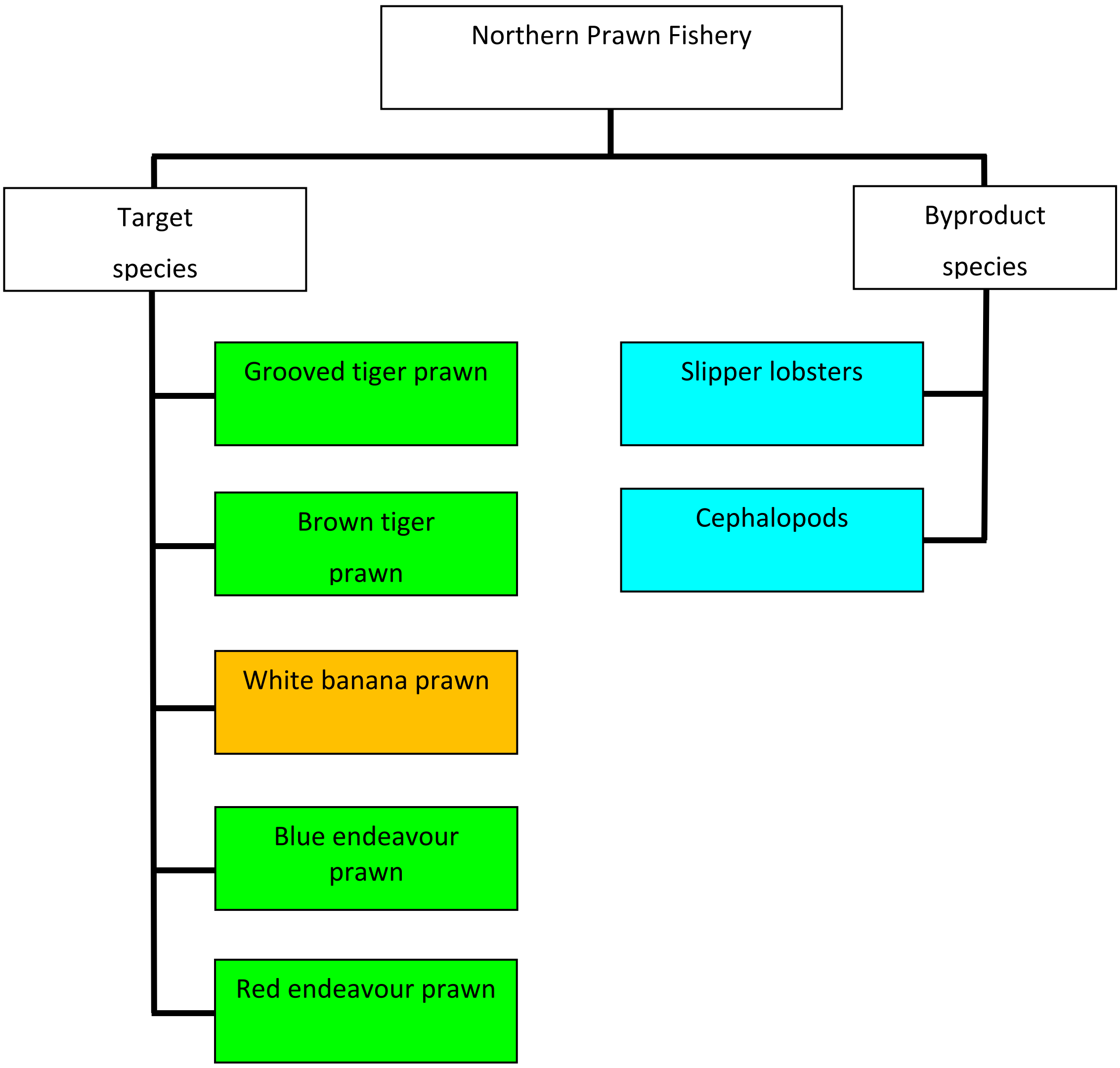

Figure 2.10. The qualitative risk assessment component tree for target and byproduct species in the Northern Prawn Fishery.

7.2 Offshore Gillnet Fishery (N12 and N13)

The important species of the GoC N12 and N13 fisheries can be divided into target and byproduct species (Figure 2.11). The target species included the principal target species, Grey Mackerel and Black-tip Shark, as well as other commonly caught species, Bull Shark and Hammerhead Sharks (three species). Two species were included as byproduct, Spanish Mackerel and Talang Queenfish.

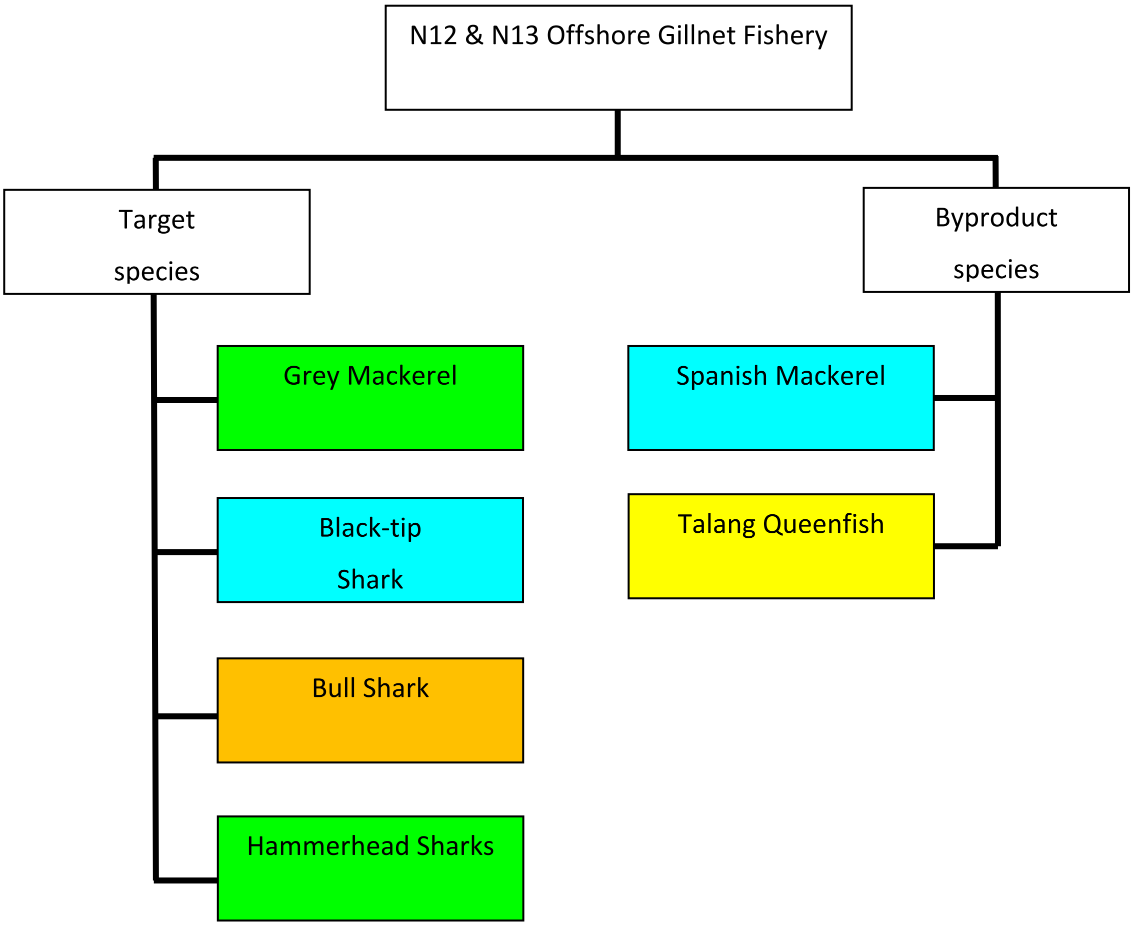

Figure 2.11. The qualitative risk assessment component tree for target and byproduct species in the Gulf of Carpentaria N12 and N13 offshore gillnet fishery.

7.3 Inshore Gillnet Fishery (N3)

The important species of the N3 fishery can be divided into target and byproduct species (Figure 2.12). The target species included the principal target species, Barramundi, King Threadfin and Blue Threadfin, as well as other important species, Grey Mackerel, Mud Crab and Black-tip Sharks. Four main byproduct species were included, Grunter, Spanish Mackerel, Talang Queenfish and Blue Catfish.

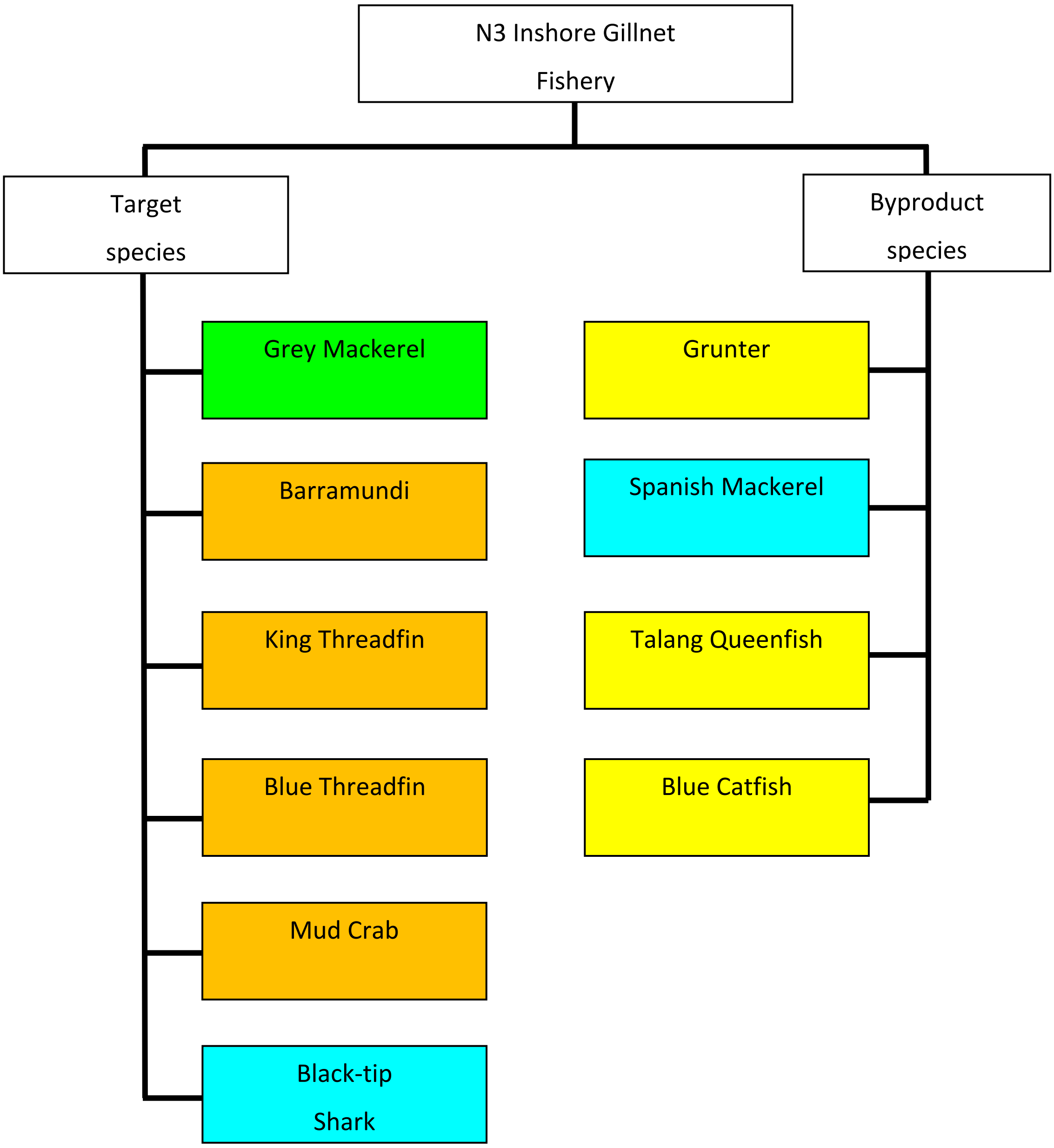

Figure 2.12. The qualitative risk assessment component tree for target and byproduct species in the Gulf of Carpentaria N3 inshore gillnet fishery.

7.4 Mud Crab Fishery (C1)

The important species of the C1 Mud Crab Fishery can be divided into a single target and byproduct species (Figure 2.13). The target species included is the fishery's principal target species, Mud Crab, while the primary byproduct species is Blue Swimmer Crab.

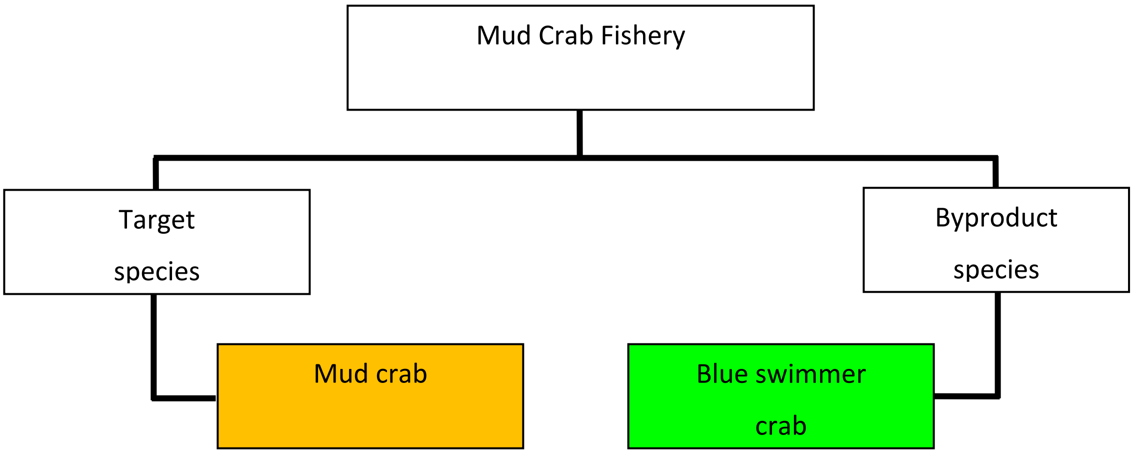

Figure 2.13. The qualitative risk assessment component tree for target and byproduct species in the Gulf of Carpentaria Mud Crab fishery.

7.5 Line Fishery (L4)

The important species of the GoC Line Fishery can be divided into target and byproduct species (Figure 2.14). The target species included the principal target species of the fishery, Spanish Mackerel. Three byproduct groups were included, Grey Mackerel, Red Snappers (five species of Lutjanids) and Black-tip Shark.

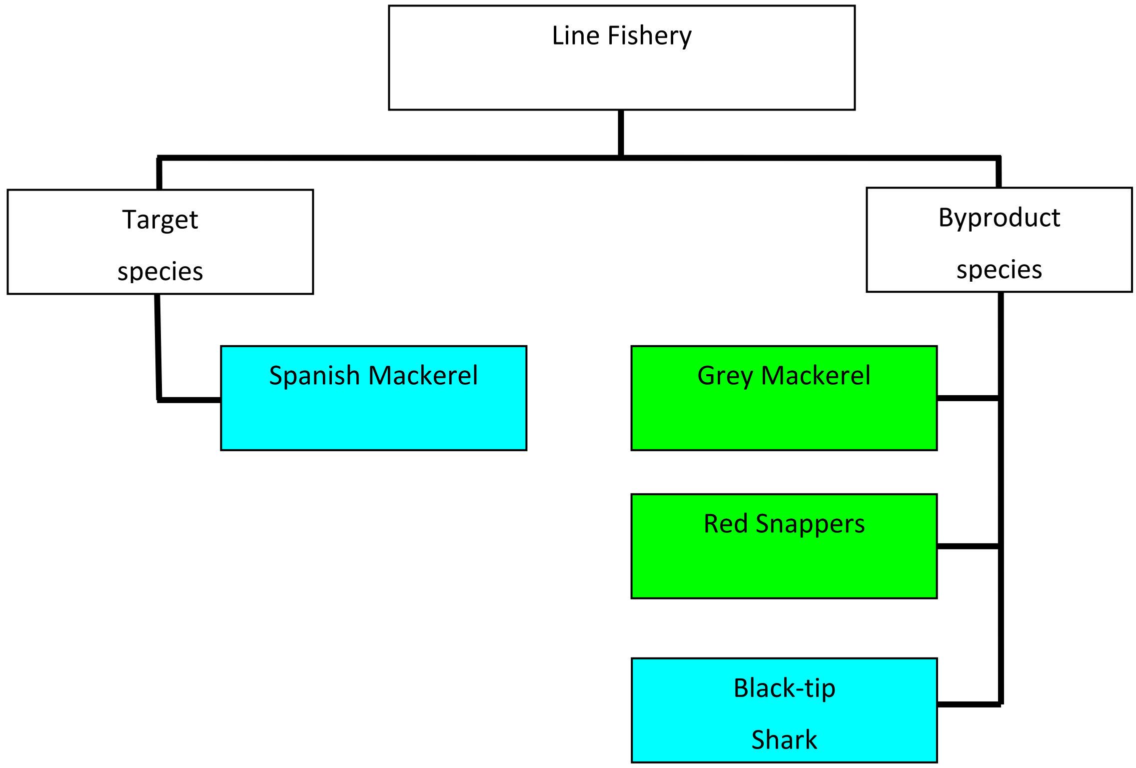

Figure 2.14. The qualitative risk assessment component tree for target and byproduct species in the Gulf of Carpentaria Line Fishery.

7.6 Developmental Fin fish Trawl Fishery

The important species of the GoC Developmental Fin Fish Trawl Fishery can be divided into target and byproduct species (Figure 2.15). The target species included the principal target species of the fishery, Crimson Snapper and Saddletail Snapper. The four byproduct species included were Mangrove Jack, Goldband Snapper, Red Emperor and Golden Snapper.

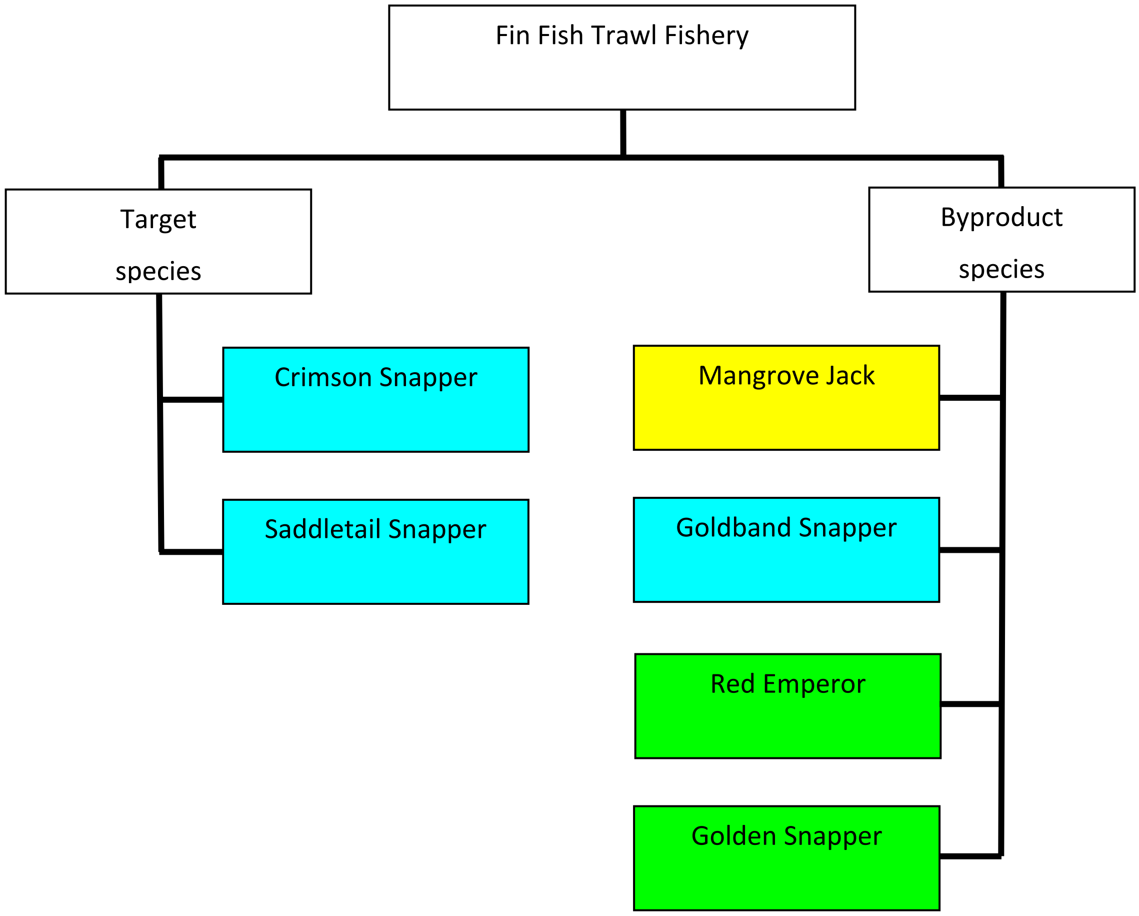

Figure 2.15. The qualitative risk assessment component tree for target and byproduct species in the Gulf of Carpentaria Developmental Fin Fish Trawl Fishery.

7.7 Indigenous Fishery

The species of importance to the Indigenous fishery can be divided into listed species and common target species (Figure 2.16). The listed species include Marine Turtles (including harvesting of eggs) and Dugong. Six other target species were included; Prawns, Mullet (primarily Liza spp.), Pikey Bream, Blue Catfish, Mud Crab, and Barramundi.

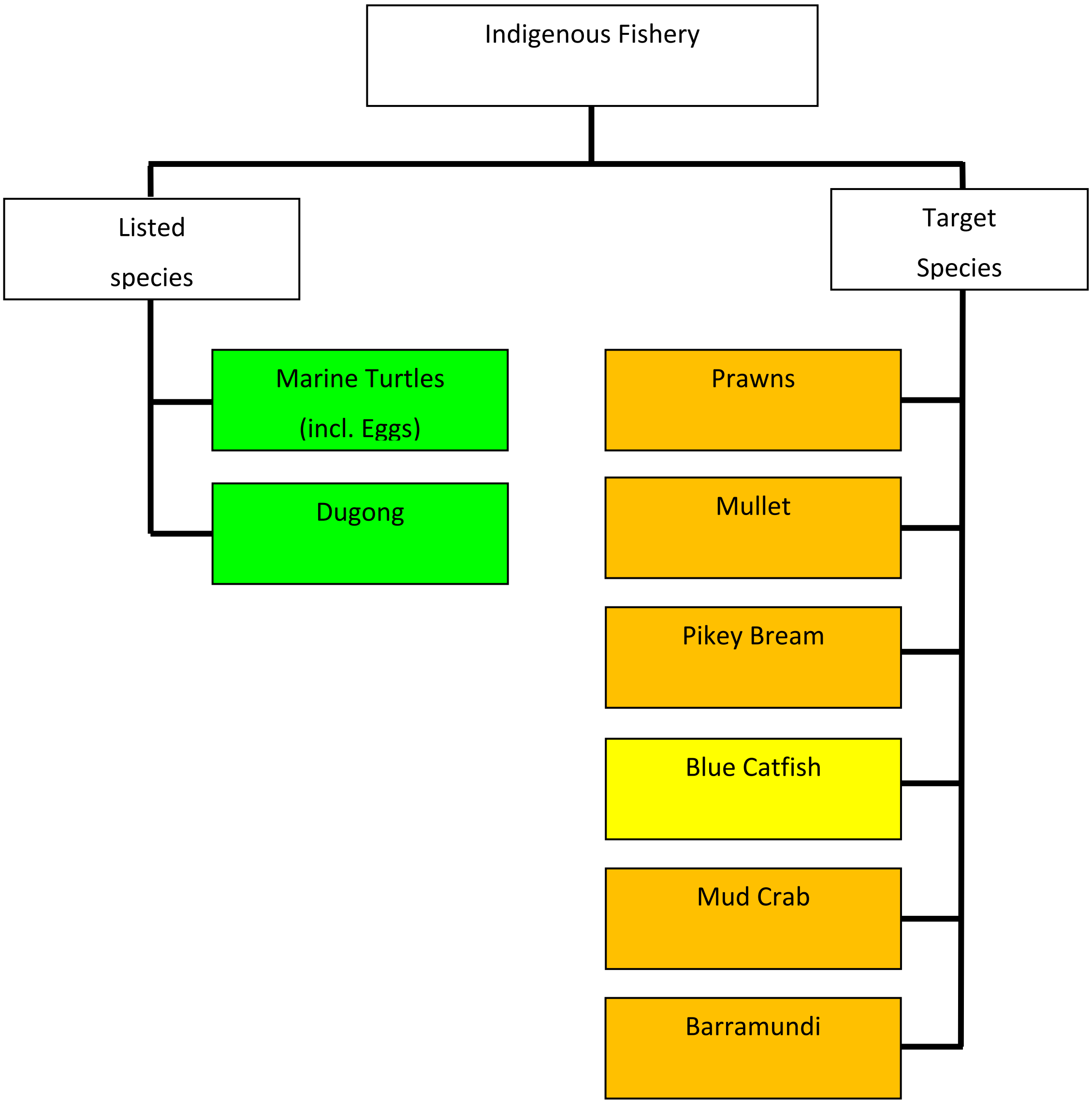

Figure 2.16. The qualitative risk assessment component tree for listed and target species in Queensland's Indigenous fishery.

7.8 Recreational Fishery

The species of importance to the recreational fishery can be broadly separated into iconic and target (Figure 2.17). The iconic species included Barramundi, Mangrove Jack, Blue Threadfin, and King Threadfin. The target species included Pikey Bream, Grunter, Mullet (primarily Liza spp.), Sooty Grunter, and Mud Crab.

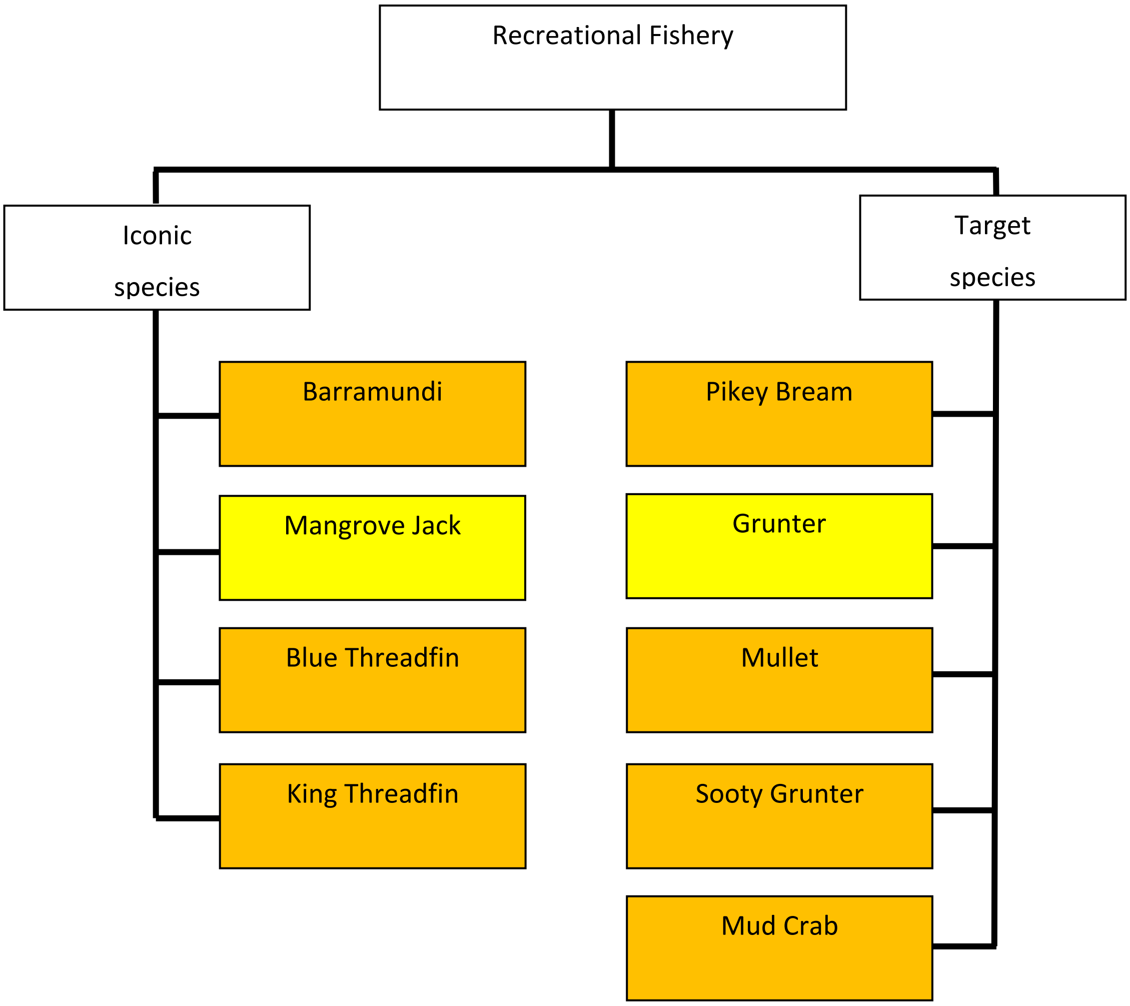

Figure 2.17. The qualitative risk assessment component tree for iconic and target species in the Gulf of Carpentaria recreational fishery.

7.9 Species of Conservation Importance

The species of conservation importance were categorised by their listing under the EPBC Act, or by the IUCN (Figure 2.18). EPBC listed species generally fell into broader taxonomic groups including Sawfish, Marine Turtles, Speartooth Shark, Dugong, Saltwater Crocodile, Migratory Shorebirds, and Sea snakes. IUCN listed species were Freshwater Whipray, Dugong, Saltwater Crocodile, and Migratory Shorebirds.

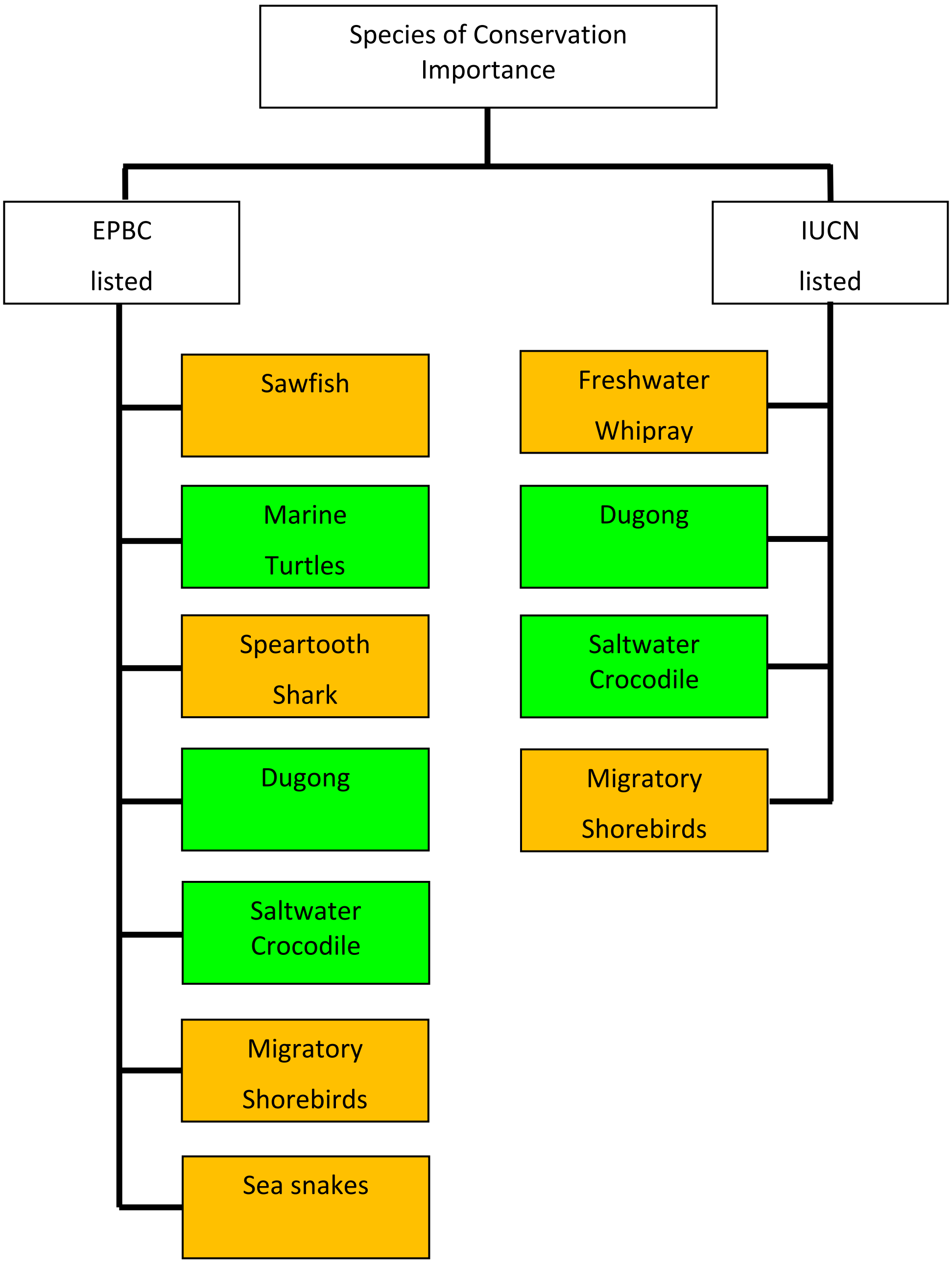

Figure 2.18. The qualitative risk assessment component tree for species of conservation importance in the southeastern Gulf of Carpentaria that are listed under the *Environment Protection and Biodiversity Conservation (EPBC) Act 1999*, or the International Union for Conservation of Nature (IUCN).

7.10 Ecosystem integrity

Species of importance for maintaining ecosystem integrity were broadly categorised by trophic level (Predator or Forage species) and trophic role (i.e. keystone) (Figure 2.19). Predator species generally included large-growing species such as Sharks and pelagic fish, while forage species included highly abundant schooling fish. Keystone species identified by the GoC ecosystem model (Griffiths et al. 2010) included a range of species from Plankton, Tiger Prawns to Bull Sharks.

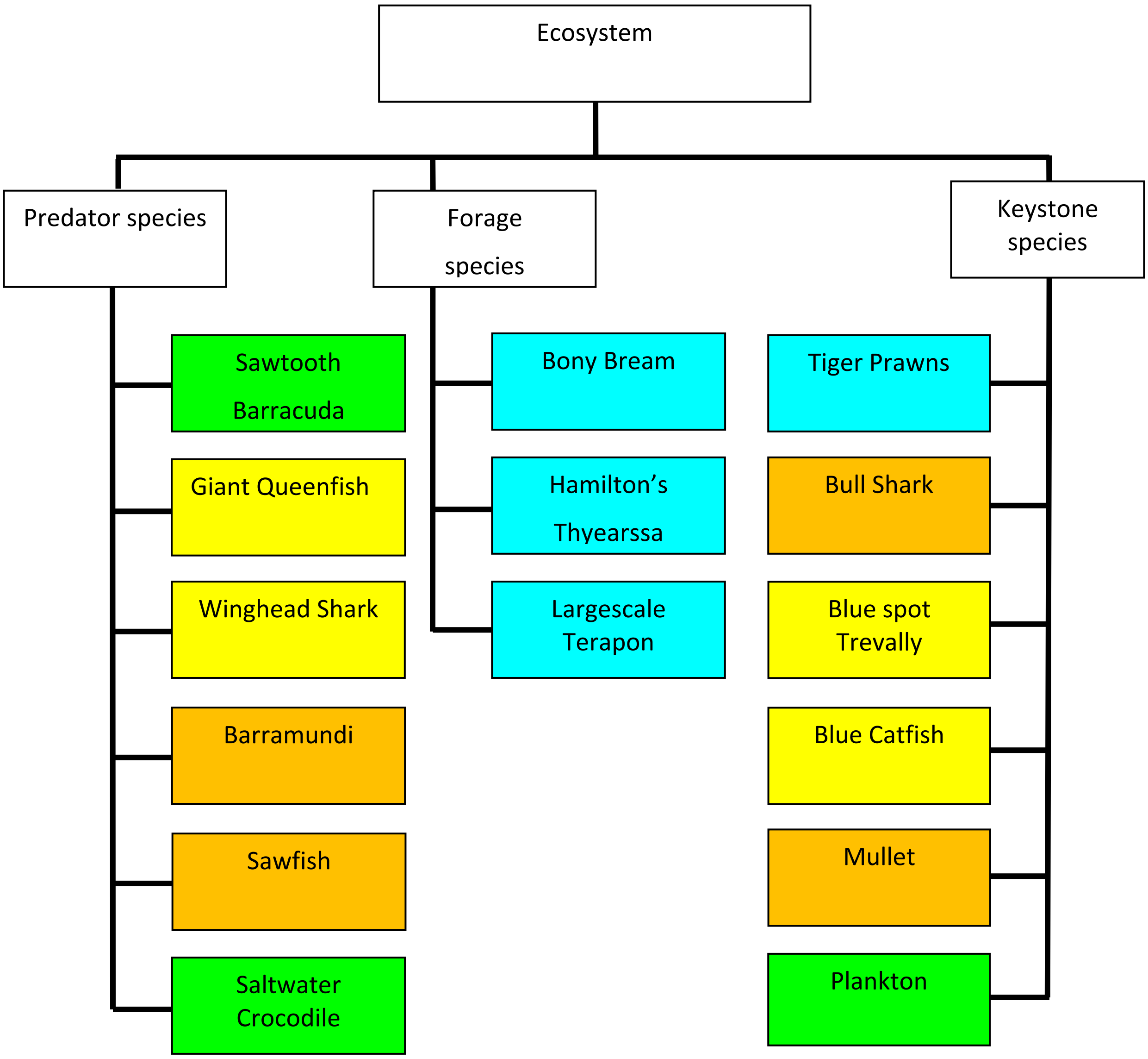

Figure 2.19. The qualitative risk assessment component tree for species considered to be of importance for maintaining the structure and function of estuarine and coastal ecosystems in the southeastern Gulf of Carpentaria.

7.11 Overall Risk Assessment Results

A total of 47 species (or species groups) and 4 habitats were assessed as to the relative risk of decline following a reduction in river flows in the Flinders and Gilbert river catchments (Table 2.13). Of these, 16 species/species groups were ranked as high risk, 8 as moderate risk, 13 as low risk, and 10 as negligible risk. With respect to habitats, Mangroves and Seagrasses were ranked as low risk, while Floodplains and Saltflats were ranked as moderate risk (Table 2.13).

Of the species ranked as high risk, Freshwater Sawfish, Speartooth Shark, Northern River Shark and Freshwater Whipray had the highest risk score overall (16) as they each were highly reliant upon estuaries and rivers to complete part, or all, of their life cycle. The lowest risk species (score 0) were those that completed their entire life cycle outside of estuaries, generally in offshore waters, had a wide geographic distribution that extended beyond the GoC, showed little evidence of population subdivision and only used estuaries opportunistically with other inshore habitats, usually as juveniles.

The following section provides a detailed literature review and a life history conceptual model for each species and habitat assessed. These reviews describe the biology, ecology and fisheries for each species in relation to reduced river flows. Each review concludes with a concise justification of the risk scores used in the qualitative risk assessment.

Table 2.13. Consequence and likelihood scores and overall relative risk rank for each species (or species group) and habitat included in the qualitative risk assessment to determine the potential impact of reduced flows in the Flinders and Gilbert rivers on the long term sustainability of their populations.

Risk ranking	Species /Group	Consequence	Likelihood
HIGH	Freshwater Sawfish	4	4
HIGH	Speartooth Shark	4	4
HIGH	Northern River Shark	4	4
HIGH	Freshwater Whipray	4	4
HIGH	Sea snakes	3	4
HIGH	Barramundi	3	4
HIGH	Pikey Bream	3	4
HIGH	Sooty Grunter	3	4
HIGH	Blue Threadfin	3	4
HIGH	King Threadfin	3	4
HIGH	Mullet	3	4
HIGH	Mud Crab	3	4
HIGH	White Banana Prawn	3	4
HIGH	Bull Shark	3	4
HIGH	Migratory Shorebirds	3	4
MODERATE	Green Sawfish	3	3
MODERATE	Narrow Sawfish	3	3
MODERATE	Mangrove Jack	2	4
MODERATE	Grunter	2	3
MODERATE	Black Jewfish	2	3
MODERATE	Talang Queenfish	2	3
MODERATE	Blue Catfish	2	3
MODERATE	Winghead Shark	2	3
LOW	Blue Swimmer Crab	1	3
LOW	Red Emperor	1	3
LOW	Golden Snapper	1	3
LOW	Grooved Tiger Prawn	1	3
LOW	Brown Tiger Prawn	1	3

Risk ranking	Species / Group	Consequence	Likelihood
LOW	Cephalopods	1	3
LOW	Grey Mackerel	1	3
LOW	Blue spot Trevally	1	3
LOW	Hammerhead Sharks	1	3
LOW	Saltwater crocodile	1	3
LOW	Dugong	1	3
LOW	Marine Turtles	1	3
LOW	Plankton	1	3
NEGLIGIBLE	Crimson Snapper	0	3
NEGLIGIBLE	Saddletail Snapper	0	3
NEGLIGIBLE	Goldband Snapper	0	3
NEGLIGIBLE	Largescaled terapon	0	3
NEGLIGIBLE	Sawtooth Barracuda	0	3
NEGLIGIBLE	Bony Bream	0	3
NEGLIGIBLE	Hamilton's Thryssa	0	3
NEGLIGIBLE	Slipper Lobsters	0	2
NEGLIGIBLE	Spanish Mackerel	0	2
NEGLIGIBLE	Black-tip Shark	0	2

Habitats

Risk ranking	Habitat	Consequence	Likelihood
MODERATE	Floodplains	2	4
MODERATE	Saltflats	2	4
LOW	Mangroves	1	4
LOW	Seagrasses	1	4

8 Literature review of assessed species and habitats

For qualitative ecological risk assessments, where risk scores can be highly subjective if they depend on stakeholder viewpoint, it is important to be as transparent as possible on how each score was generated (Fletcher, 2005). This section provides a review of the literature on the biology, ecology and fisheries for each species (or species groups e.g. Marine Turtles) included in the qualitative risk assessment to provide a basis for the reader to understand how each species may be affected by altered river flows in the Flinders and Gilbert river catchments. At the end of each species review the 'risk score justification' section details the key points from the review that formed the basis of the consequence and likelihood risk scores.

8.1 Barramundi (*Lates calcarifer*)

Barramundi (*Lates calcarifer*) is distributed widely throughout the west Indo-Pacific region, including rivers, lagoons, swamps and estuaries across northern Australia from the Noosa River in Queensland to the Ashburton River in Western Australia (Froese and Pauly, 2014). Barramundi typifies a life history strategy that could be significantly impacted by interruptions to the natural flows of the Flinders and Gilbert rivers.

Barramundi demonstrates a catadromous (non-obligatory) life history strategy. Spawning occurs in the lower estuary and adjacent coastal zone before the on-set of the wet season but can extend between September to February. Their downstream movement to these areas is possibly stimulated by rising water temperature; increasing photoperiod and first-season low flows that connect riverine waterholes and reduce salinity in the upper estuary (Robins and Ye, 2007). Larvae spend about three weeks in inshore marine waters and brackish waters to optimise their development (Robins and Ye, 2007). Although juvenile Barramundi can survive as permanent estuarine residents, they thrive in semipermanent wetlands, tidal creeks and freshwater riverine habitats (Russell and Garrett, 1985). Postlarvae and small juveniles attempt to access freshwater habitats adjacent to, and upstream of, the estuary (Halliday et al., 2012).

Barramundi recruitment to nursery habitats is moderated by floodwater access to supra-littoral, lagoon and riverine habitats. Both longstream and floodplain connectivity require significant flood heights that allow fish to travel upstream or out of the river channel in search of habitats that increase their survival and growth during their juvenile stage. Peak spring tides also may facilitate access to supra-littoral habitats, supplemented by small early-season floods. Juvenile and adolescents remain in ephemeral/perennial freshwater habitats from months to years until flood-moderated connectivity liberates them to return to the river before emigrating downstream to the estuary and nearshore zones, often as adults to spawn. The annual wet season and subsequent runoff is a major determinant of their access to juvenile habitat and connectivity back to the coastal zone. There is a correlation between seasonal floodflow and juvenile recruitment strength and subsequent adult stocks, possibly lagged by one to five years (Robins et al., 2005; Halliday et al., 2012).

Barramundi is a protandrous hermaphrodite, changing sex from males as smaller individuals to females at larger sizes. Therefore, the sex ratio and successful spawning of the estuarine population may be interrupted by floodplain/riverine connectivity in that males in freshwater habitats may not be able to move downstream. Although some fish migrate large distances, most remain and show high fidelity to a small number of estuaries in a region (Robins and Ye, 2007). This provides potential for subdivision of the GoC population. A series of dry years when tropical Australian rivers cease to flow and barriers built as water infrastructure both prevent juvenile recruitment to freshwater habitat and long-stream movement. It is postulated that over consecutive dry years, with limited or no access to freshwater habitats, the population of barramundi declines. Flood flows may stimulate primary productivity in freshwater and estuarine habitats that enhances their populations (Robins et al., 2006).

The interruption of first-season floods by in-stream dams and the reduction of both low and high floodflows by water diversion or impoundment may have a negative effect on both the downstream movement of adult Barramundi to spawn, and the habitat access, growth and mortality of juvenile fish

(Robins and Ye, 2007). Physical barriers to longstream and cross floodplain connectivity such as in-stream dams and barrages and landscape modification interrupt Barramundi movements to freshwater habitats and their return to the estuary/nearshore. These effects will likely impact upon the fishery catch.

Barramundi shows high plasticity in growth rates dependent of their local environmental conditions (Robins et al., 2006; Robins and Ye, 2007). During wet years, their growth and the juvenile habitat available to Barramundi increases markedly and the population, including the estuarine population, increases (Halliday et al., 2012). This may be an indicator of population subdivision. Barramundi in the south-east GoC from the Mitchell River to the Leichardt River comprise the same genetic stock, but are distinct from stocks in the NT and the north-west and east coasts of Cape York (Shaklee and Salini, 1985; Keenan, 1994; Jerry et al., 2013).

Barramundi is arguably the most important species to commercial, recreational and Indigenous fisheries throughout the GoC. Barramundi have been fished commercially in the GoC since the 1970s and are fished from February to September each year. Barramundi has been the most important commercial species (by weight) over the past two decades, with catches steadily increasing from around 400 t in the mid-1990s to a peak of 977 t in 2011 (Figure 2.20), whereas the annual mean catch over the past five years is 793 t. Recent fishing effort has been relatively stable at about 7500 days per year between 2007 and 2012 (Jerry et al., 2013). Although data are not available by fishery, most of the catch comes from the N3 inshore gillnet fishery, and a small proportion comes from the N12/13 offshore gillnet fishery. After 2000, all Barramundi was caught by the N3 inshore gillnet fishery.

Barramundi is an iconic sport and table fish for recreational fishers in the GoC and is probably the most important species to this fishery. It is particularly important in the south-east GoC during the annual dry season, when road access to townships is open for tourists travelling from both interstate and from within Queensland. The 2010 statewide recreational fishing survey estimated the total catch in the southeast GoC to be 125,000 fish (Table 2.5). Using an average weight of 5.17 kg (unpublished data, James Webley, DAFF), this equates to a total catch biomass of around 646 t. However, unpublished data from the 2010 survey indicates that 83% of Barramundi in the region are released (QFISH online database). Logbook catches for Barramundi in the charter fishery peaked at around 2 t in the early 2000s, but the average reported retained catch in the past five years was only 470 kg.

The Indigenous catch of Barramundi in the study region is less certain than other fisheries. The only available data were collected for all of Queensland as part of the 2001 national recreational and Indigenous fishing survey (Henry and Lyle, 2003), which was an estimated catch of 5745 fish, or 29.7 t.

Risk score justification

The species is dependent on estuaries to complete its life cycle and uses marine to freshwater habitats (Figure 2.21). The highly variable growth rates documented between rivers suggest the population may be highly subdivided to the extent that the Flinders and Gilbert rivers may support individual populations. Coupled with its longevity (~20 years) and protandrous sex change, these life history traits suggest that individual populations may be vulnerable to decline, especially if connectivity between key habitats is compromised due to reduction in flow.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

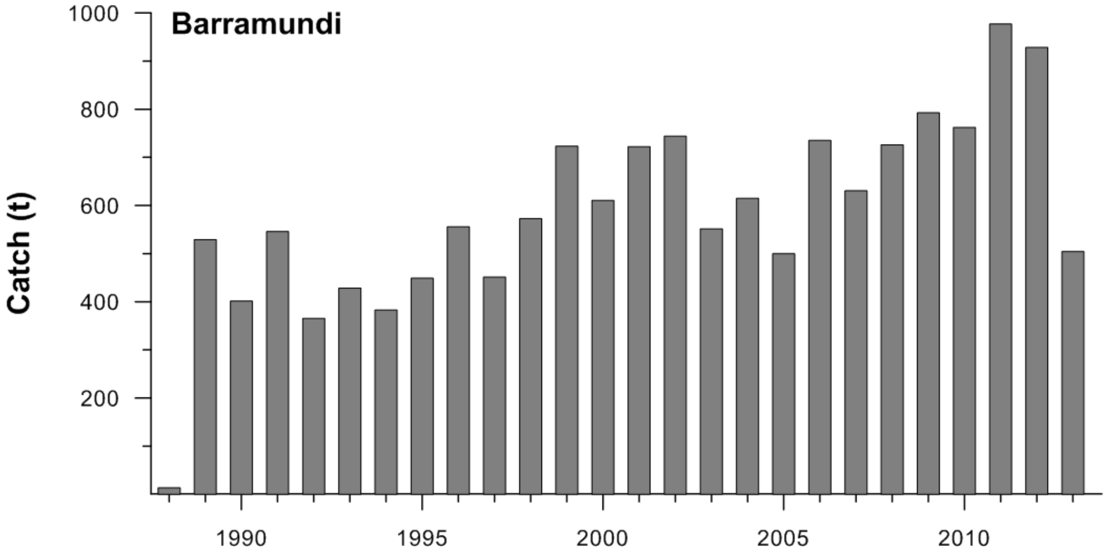

Figure 2.20. Annual catch of Barramundi (*Lates calcarifer*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

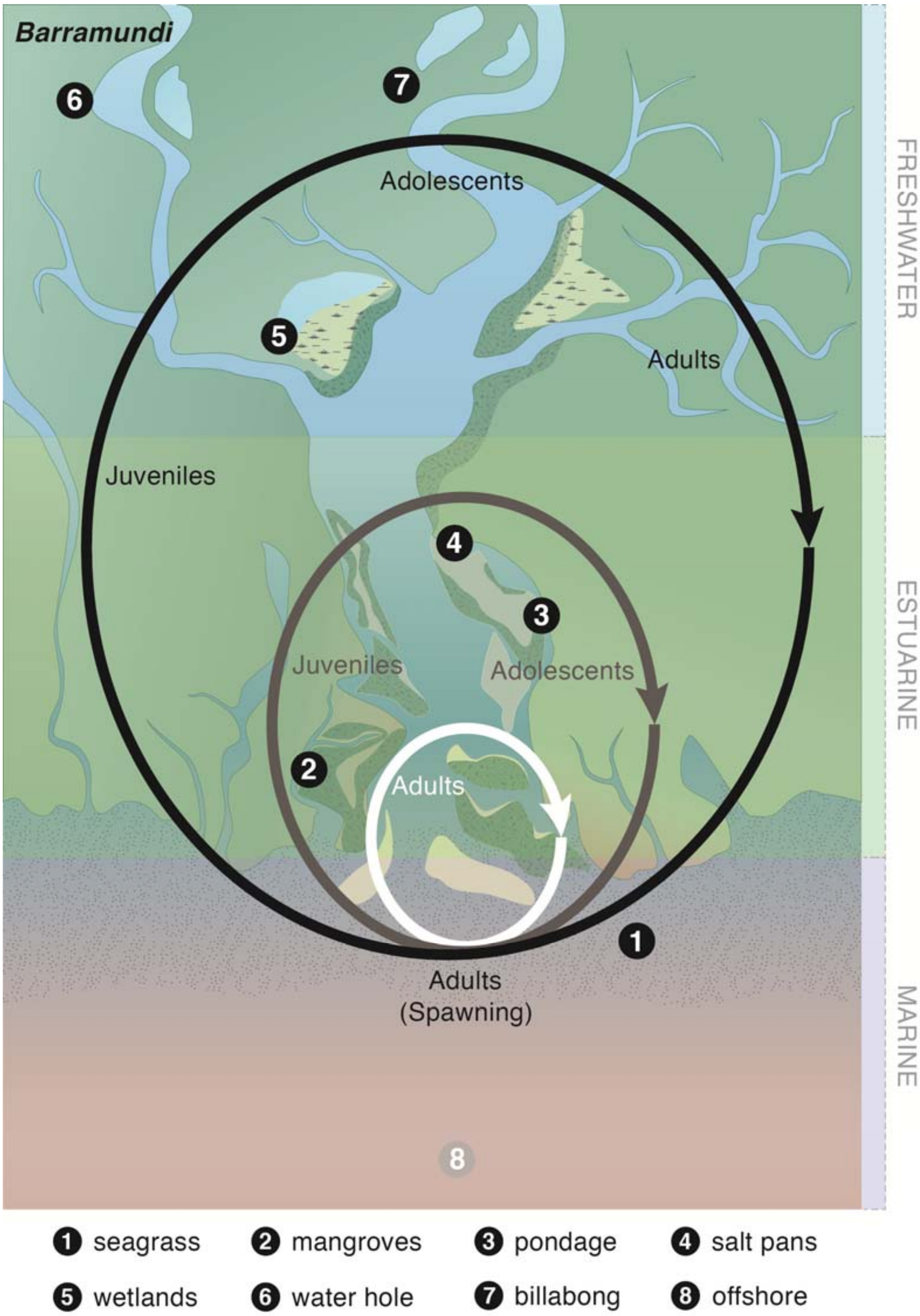

Figure 2.21. Conceptual model of the life history of Barramundi (*Lates calcarifer*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.2 Mangrove Jack (*Lutjanus argentimaculatus*)

Mangrove Jack (*Lutjanus argentimaculatus*) is a significant component of GoC fisheries and has a life history strategy that may be impacted by interruptions to the natural flows of the Flinders and Gilbert rivers. Mangrove Jack has a larval life history strategy, spawning offshore in structurally-complex reef or deepwater habitats in October to February after females fish mature at 512 mm length to caudal fork (LCF) (Russell et al., 2003). Its eggs hatch and postlarvae and small juveniles advect inshore to access estuarine and freshwater habitats in the lower rivers reaches (Russell et al., 2003; Russell and McDougall, 2005). Freshwater cues from coastal estuaries may orientate recruiting juveniles to move inshore (Robins and Ye, 2007).

Along the Queensland coast, small juvenile Mangrove Jack (<100 mm LCF) are found in estuaries from February to October, though their abundance tends to be high from February to June (with a few >80 mm in November to January). Juveniles and adolescents are euryhaline, capable of inhabiting waters from near-fresh to marine salinity, 1 to 35 Practical Salinity Units (PSU) (Russell et al., 2003; Robins and Ye, 2007). They are abundant throughout tropical and sub-tropical estuaries, and in east Qld coast rivers are in highest abundance within the first 10 km from estuary mouths (Russell and McDougall, 2005). Juvenile Mangrove Jack are found in ephemeral and perennial freshwater habitats in the lower reaches of the rivers and near 100% of fish <300 mm LCF are found in freshwater habitats (Russell and McDougall, 2005). For shelter, juvenile Mangrove Jack rely on structured habitat such as snags, rocky bank areas and mangrove roots. The annual wet season and subsequent runoff is a major determinant of their access to juvenile habitat and connectivity back to the coastal zone. Tagging studies showed that about two-thirds of juvenile fish move less than 1 km from their release location until they start an ontogenetic migration offshore. About 8% moved up-river and 1% made inter-riverine movements (Russell et al., 2003; Russell and McDougall, 2005).

The recruitment of postlarval and juvenile Mangrove Jack to freshwater nursery habitat is moderated by floodwater access to supra-littoral, lagoon and riverine habitats. Both longstream and floodplain connectivity requires significant flood heights that allow fish to travel up to 100 km upstream (Russell et al., 2003) or out of the river channel in search of habitats that increase their survival rate and growth during their juvenile stage. Peak spring tides from December to February also may facilitate access to supralittoral habitats, supplemented by small early-season floods. Mangrove jack can remain in permanent off-stream freshwater habitats for years (e.g. a 610 mm fish nine years old was found by Russell and McDougall, 2005) and in a reproductively immature or 'resting' state until flood-moderated connectivity liberates them to return to the estuary and then migrate offshore, eventually to spawn.

Mangrove Jack benefit from ontogenetic migrations; from offshore to inshore as larvae and juveniles to access littoral and freshwater refuge and foraging habitats; and from inshore to offshore as adolescents. Mortality of small fish in floodplain and supra-littoral habitats is lower than in stream channels. Similarly, the condition of small fish (400–500 mm LCF) in estuaries was better than fish caught in either freshwater or offshore habitats (Russell and McDougall, 2005). These finding suggests that the survival and growth of juvenile to adolescent fish is enhanced in estuarine conditions, so connectivity mediated by floodflow between freshwater and estuarine habitats may affect recruitment to the offshore fishery. Mangrove Jack show high levels of gene flow across tropical Australia and are probably the same genetic stock (Russell et al., 2003).

Mangrove Jack is important to commercial, recreational and Indigenous fisheries in the GoC. The species is almost exclusively caught in the finfish developmental trawl fishery where catches peaked at 74 t in 2005 but have declined to 3 t in 2012 (Figure 2.22). It is unknown whether this decline in catch is due to a decline in the population of fish, or is an artefact of reduced fishing effort.

Mangrove Jack is an iconic recreational species across northern Australia, being taken in the charter and recreational fishery in estuaries and coastal habitats generally during the dry-season. Although the charter catch is low in the GoC, averaging only 180 kg in the past five years, the recreational catch is more substantial. The 2010 statewide recreational fishing survey estimated 4000 fish were caught (Table 2.5), or approximately 4 t assuming an average catch weight of 1 kg.

Although not mentioned specifically, Indigenous fishers favour estuarine locations in the GoC as fishing spots and logically Mangrove Jack would be caught (Barber, 2013).

Risk score justification
Mangrove Jack can complete most of its life cycle outside of estuaries and resides and spawns offshore as adults. However, a large proportion of the GoC population uses estuarine and freshwater habitats, particularly as juveniles (Figure 2.23). The interruption of first-season floods by in-stream dams and the reduction of both low and high floodflows by water diversion or impoundment may have a negative effect on supra-littoral habitat access, growth and mortality of juvenile fish and the downstream and cross-wetland movement of adolescent Mangrove Jack to emigrate.

Risk scores: Consequence **2**; Likelihood **4**. **Overall risk rating**: MODERATE

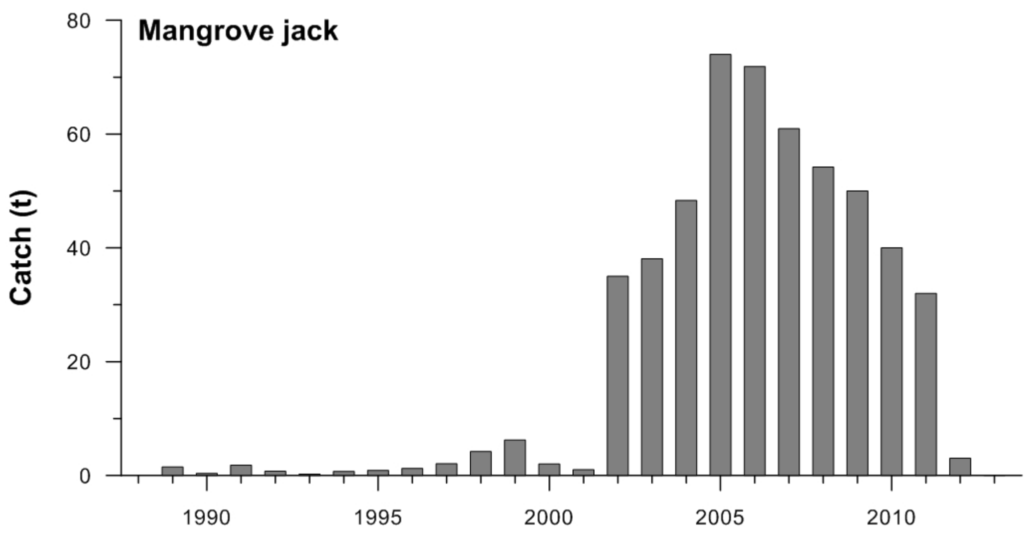

Figure 2.22. Annual catch of Mangrove Jack (*Lutjanus argentimaculatus*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

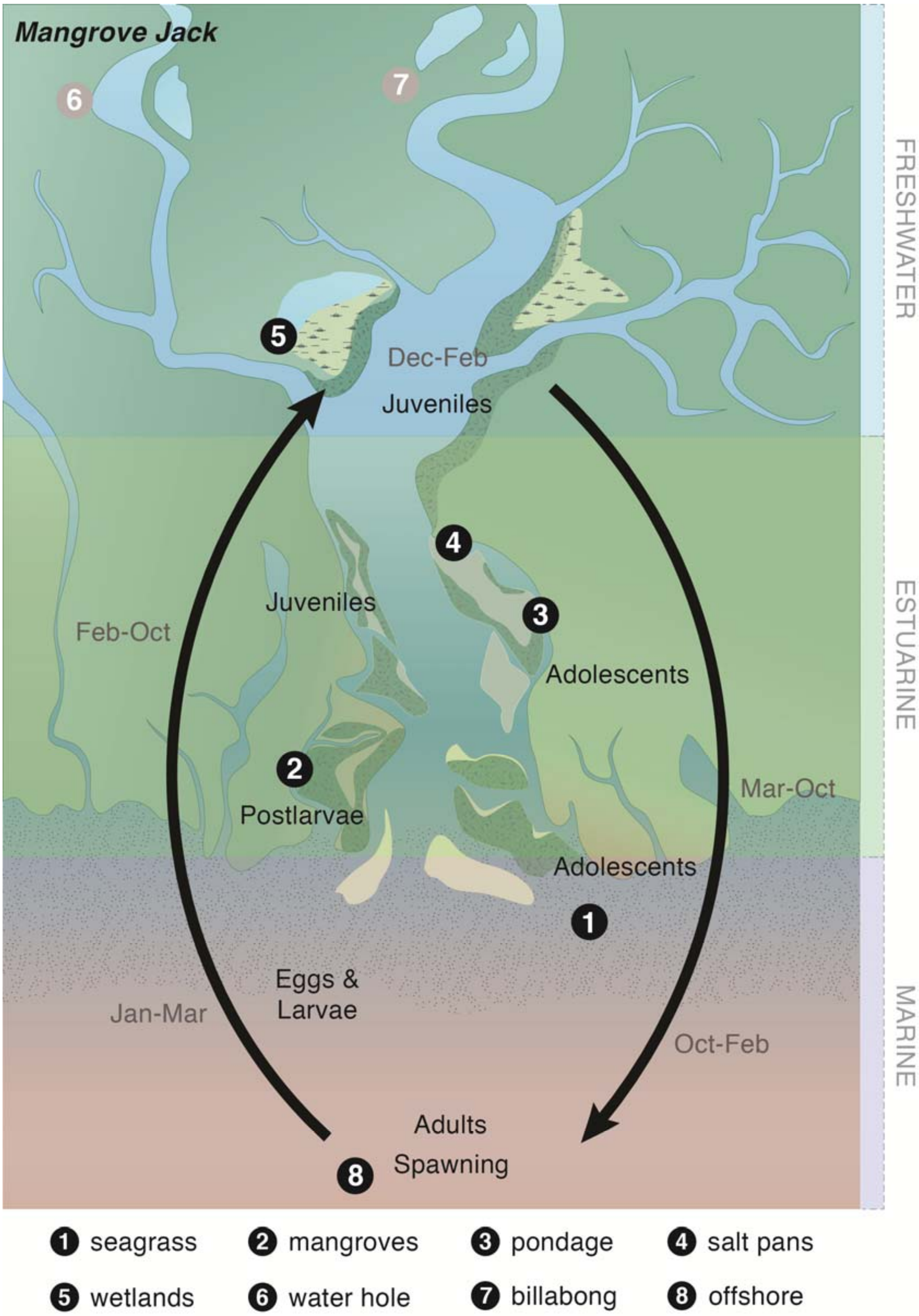

Figure 2.23. Conceptual model of the life history of Mangrove Jack (*Lutjanus argentimaculatus*) illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.3 White Banana Prawn (*Penaeus merguiensis*)

The White Banana Prawn (*Penaeus merguiensis*) typifies a life history strategy that would be significantly impacted by interruptions to the natural flows of the Flinders and Gilbert rivers (Robins and Ye, 2007). It has a larval life history strategy: spawning in the marine offshore zone during August to December, its eggs hatch to larvae that are advected inshore, where they settle as benthic postlarvae in estuarine mangrove habitats (Rothlisberg et al., 1985). They remain in the estuary for several months throughout October to February before emigrating to the nearshore and offshore zones as sub-adults.

White Banana Prawn has been fished in the GoC since the 1960s and supports one of Australia's most valuable fisheries, the NPF, where a large percentage of the catch comes from the southeast GoC. This species is fished from April to June each year (Barwick, 2013). The annual catch in reporting Zone 8 – comprising the Flinders and Gilbert rivers – is highly variable ranging from just 32 t in 1990, to peaks of 5461 t and 2818 t in 1974 and 2011, respectively (Figure 2.24). White Banana Prawn is not of significant importance to other commercial, charter, or recreational fisheries in the region. Although not specifically recorded as being caught by these fisheries, anecdotal accounts suggest that recreational fishers opportunistically capture the prawns in cast nets as they emigrate from estuaries in aggregations. However, these catches would probably be less than a few hundred kilograms annually. In contrast, prawns (most likely White Banana Prawns) were the most abundant species caught by Indigenous fishers in the 2000/01 NRIFS, where an estimated 131,158 prawns were caught (Table 2.6).

Simultaneous with the development of the commercial fishery, over 50 years of scientific research has explored the behaviour, habitats, food, growth, ontogenetic movement and predation upon each stage of the life history of the White Banana Prawn (Staples, 1980b; 1980a; Somers and Kirkwood, 1984; Rothlisberg et al., 1985; Vance and Staples, 1992; Robins et al., 2005). Gulf-wide, the catch of White Banana Prawn is strongly related to rainfall and subsequent runoff (Buckworth et al., 2013); rainfall is the driver of the often more than threefold variability in catch.

The relationship between White Banana Prawn catch and rainfall was evident from the 1970s when research in the estuary of the Norman River showed a strong emigration response in juvenile prawns when floodflows caused the salinity of estuarine water to drop to low levels (Staples, 1980a). To this day the management of the GoC White Banana Prawn fishery recognises the environmental drivers determining catch; managing sustainability by limiting the number of vessels licensed to fish and the closure of the fishing season using a catch rate trigger (Zhou et al., 2014). This contrasts with stock assessment-based management of other GoC prawn species for which a strong annual spawning stock and stable habitats sustain subsequent catch (Punt et al., 2011).

The relationship between floodlfows and White Banana Prawn catch is particularly strong in the southeastern GoC where ~70% of the variation in catch offshore of the Staaten, Gilbert, Norman, Flinders, Leichardt and Nicholson rivers can be explained by rainfall in the river catchments (Staples and Vance, 1985; Venables et al., 2011). There is a simple physiological response of juvenile White Banana Prawns that explains their emigration in response to a freshwater cue (Dall et al., 1990). However, during large flows primary production and food availability in the estuary also decline and reduce the habitat value of the estuary for the White Banana Prawn population (Burford et al., 2011). Importantly, large floodflows transfer hundreds of tonnes of nutrients (phosphorus and nitrogen) to the nearshore flood plume zone which may stimulate an enriched food web that supports the abundant population of sub-adult White Banana Prawns that have recently emigrated (Burford et al., 2011).

When juvenile White Banana Prawns are resident in the estuary they also benefit from low floodflows that create brackish conditions in the estuary. The growth and mortality of White Banana Prawns is optimal in brackish waters characteristic of the tropical estuaries; about 28°C and a salinity of 25 (Staples and Heales, 1991). Estuaries in the southern GoC become hypersaline in the lead up to the wet season and in years of very low rainfall. Under hypersaline conditions, growth of the prawns may be inhibited until first rains and low flows reduce the salinity in the tropical estuaries to brackish levels. Typical field-measured growth rates of juveniles in GoC estuaries range from 0.63 to 1.65 mm carapace length (CL) wk^{-1} (Haywood and Staples, 1993) and in the Fitzroy River White Banana Prawn growth was positively influenced by rainfall and floodflows (Robins et al., 2005). The mortality of estuarine juveniles is high at 0.89 wk^{-1} for 2 mm CL prawns to 0.02 wk^{-1} for 15 mm prawns (Wang and Haywood, 1999). Offshore, their natural mortality in the southeastern GoC is lower, estimated at 0.046 wk^{-1} in the 1970s, and at 0.03 wk^{-1} most recently (Zhou et al., 2014) as an average for all sizes of adult prawns. An overview of the life history of the species is provided in Figure 2.25.

Thus, the interruption of first-season floods by in-stream dams and the reduction of both low and high floodflows by water diversion or impoundment will have a negative effect on the growth, mortality and the emigration of juvenile White Banana Prawns. A secondary effect of the use of water for irrigation is the change in the baseflow characteristics below the irrigated farms. The Ord River Dam and associated irrigation infrastructure has increased the dry season flows downstream of the farmland by 439%; creating a perennial stream (Pusey et al., 2011a). The constant input of freshwater into the Ord River Estuary has transformed the upper estuary into freshwater habitat, too fresh for juvenile White Banana Prawn habitat. During the estuarine recruitment season, the abundance of juvenile prawns in the Ord estuary is much less than adjacent estuaries in Cambridge Gulf (Kenyon et al., 2004). A similar barrier to estuarine recruitment would occur in GoC rivers if irrigation facilitated perennial flows that caused estuaries to become fresh during the juvenile prawn recruitment period (September to February).

Risk score justification
There is strong evidence from quantitative field studies that demonstrates the importance of large floodflows to the White Banana Prawn. Flows are the key cue for adult prawns to emigrate from estuaries to spawn in offshore waters. Although White Banana Prawn exist as a single population throughout the GoC, the rivers of the southeastern GoC are likely to make a substantial contribution to the production of the GoC population. Consequently, a reduction in flow may reduce the number of recruits to the spawning population and affect the population size in the region.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

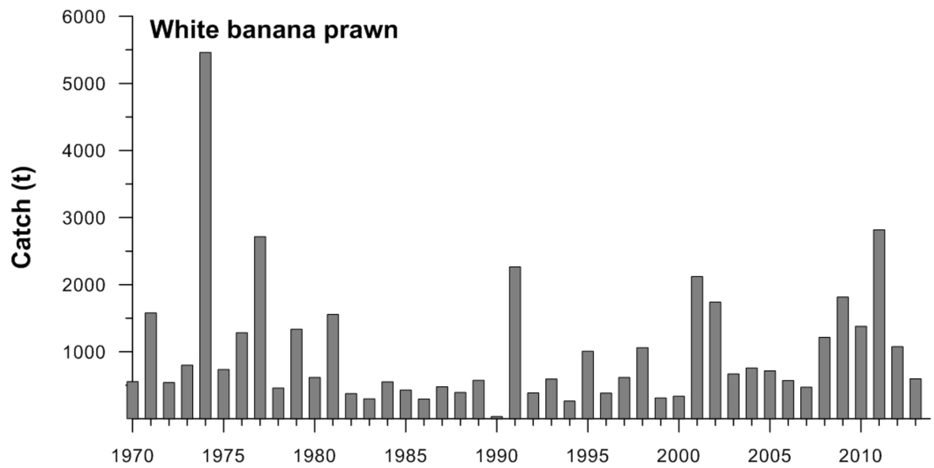

Figure 2.24. Annual catch of the White Banana Prawn (*Penaeus merguiensis*) by the Northern Prawn Fishery in catch reporting Zone 8 in the Gulf of Carpentaria. Data supplied by the CSIRO.

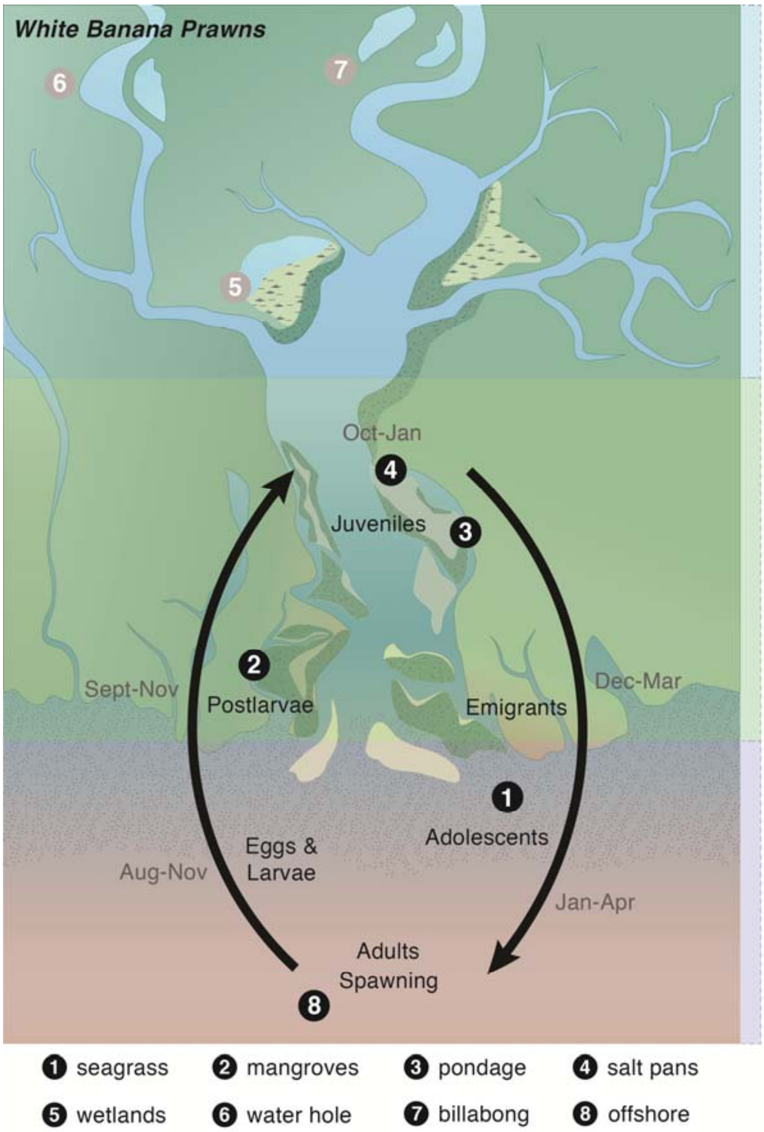

Figure 2.25. Conceptual model of the life history of the White Banana Prawn (*Penaeus merguiensis*) illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.4 Tiger and Endeavour Prawns

The Grooved and Brown Tiger Prawns (*Penaeus semisulcatus* and *P. esculentus*) and the Blue-tailed and Red-tailed endeavour prawns (*Metapenaeus endeavouri* and *M. ensis*) are species that support a major GoC fishery (Barwick, 2013). The fishery complements the Banana Prawn fishery, though temporally offset and is undertaken in the last seven months of the year. These four species typify a life history strategy that would probably not be significantly impacted by interruptions to the natural flows of the Flinders and Gilbert rivers (Dichmont et al., 2003; Punt et al., 2011). Tiger and Endeavour prawns have a near-identical larval life history strategy to White Banana Prawns: they spawn in the marine offshore zone from August to February (Rothlisberg and Jackson, 1987; Rothlisberg et al., 1987; Condie et al., 1999), their larvae are advected inshore where they settle as benthic postlarvae in coastal, embayment and estuarine seagrass habitats (Dall et al., 1990). They remain in these shallow habitats for several months before emigrating to the nearshore and offshore zones as subadults. They are fished during June and from August to November each year (Dichmont et al., 2003; Barwick, 2013). The critical difference in life history strategy is their use of submerged vegetation, mostly seagrass, in shallow coastal locations, and if in an estuary, usually further downstream than most mangrove habitats. Consequently, they are less impacted by flows. The annual wet season and subsequent runoff is not a major determinant of their emigration and catch. In fact, turbid floodflows and plumes may be detrimental to their seagrass habitats via the exclusion of sunlight needed for photosynthesis (Preen, 1995; Campbell and McKenzie, 2004).

Tiger and Endeavour prawns have been fished in the GoC since the 1970s and their catches are far less stochastic than White Banana Prawns. Endeavour prawns are caught simultaneously with Tiger Prawns, but as an incidentally catch. Targeting an un-fished stock, the Tiger Prawn catch in the entire NPF peaked at 7000 to 8000 t in the 1980s (accompanied by over allocation of effort) and declined until the mid-2000s as effort was reduced and the stock was sustainably fished (Dichmont et al., 2003; Barwick, 2013). The Endeavour prawn catch peaked just over 2000 t in the early 1980s and recently has been <500 t. Unless the stock and effort in the fishery is managed, the stocks of both Tiger and Endeavour prawns are possibly subjected to overfishing and currently a bi-annual assessment estimates the optimal economic yield (Punt et al., 2010). Most recently the Tiger Prawn fishery has operated at S_{MSY} and E_{MSY} targets (Dichmont et al., 2003; Punt et al., 2010; 2011); catch has trended towards 2000 t per year and may improve as the benefits of reduced effort (and a E_{MEY} target) support improved catches (Dichmont et al., 2008; Punt et al., 2011).

In Zone 8 of the NPF, spanning the Gilbert and Flinders river mouths, the combined catch of the four species has decline significantly from 1052 t in 1998 to a low of 40t in 2004, and an average annual catch of 227 t in the past five years (Figure 2.26). Tiger and endeavour prawns are not significantly important to other commercial, charter, or recreational fisheries in the region. The species have not been specifically recorded in these fisheries, but recreational and Indigenous fishers most likely catch small amounts opportunistically.

Simultaneous with the development of the fishery, over 30 years of scientific research has explored the behaviour, habitats, food, growth, ontogenetic movement and predation on each stage of the life history of Tiger Prawns (Staples and Vance, 1985; Crocos, 1987a; Loneragan et al., 1989; Crocos and Van der Velde, 1995; Kenyon et al., 1995; 1997; Haywood et al., 1998; Loneragan and Bunn, 1999; Dichmont et al., 2003; 2008; Punt et al., 2011). Gulf-wide, the catch of Tiger and Endeavour prawns is geographically related to the distribution of their inshore juvenile habitat and the deployment of sustainable effort in the fishery (Dichmont et al., 2003). From 1970 to 1991, the proportion of Brown and Grooved Tiger Prawns in the catch from the GoC was estimated to be 51:49%. From 1970 to 1991, the average catch of Blue- and Red-tailed endeavour prawns was 76%:24%.

In the GoC, Brown Tiger Prawns spawn throughout the year, with peak spawning occurring from August to September (Crocos, 1987a). Grooved Tiger Prawns exhibit a peak in spawning in August through October, and a minor peak in January to February (Crocos, 1987b). Both species are highly fecund and large females may produce 500,000 eggs. The minimum size at first maturity is 25 mm CL and 29 mm CL for Brown and

Grooved Tiger Prawns, respectively (about 5-7 months old) (80% inseminated, 32-50 mm CL and 38-54 mm CL, respectively). In the Gulf of Carpentaria, Blue-tailed endeavour prawns spawn throughout the year, with peak spawning occurring from September to December in the west, and July to September in the east. The spawning of Red-tailed endeavours is concentrated during September to November (Crocos et al., 2001). Both species are highly fecund. In Albatross Bay, the size at first maturity is 23 mm CL and 21 mm CL for Blue-tailed and Red-tailed endeavour prawns, respectively. The proportion of mature females increases with size (80% inseminated at 33 and 38 mm CL, respectively) (Crocos et al., 2001).

Adult Tiger and Endeavour prawns spawn at sea, usually in waters < 50 m deep. Their eggs are shed into the water column and nauplii hatch after about one day. They develop through pelagic protozoeal and mysis stages and reach near-shore waters about two to three weeks after hatching (Rothlisberg and Jackson, 1987; Rothlisberg et al., 1987). Larval behaviour is cued to the diel cycle, moving into the water column at night to feed. As they grow, postlarvae become tidally clued, moving into the water column on the flood tide to facilitate advection to inshore nursery habitat (Vance and Pendrey, 2008). Only those individuals from eggs spawned in the advective envelope relative to coastal habitats move inshore and successfully recruit to juvenile habitats (Condie et al., 1999). As they develop, Tiger Prawn postlarvae (approx 1.2 mm CL) resemble small prawns and grow to become demersal at about 1.7 mm CL. Postlarvae recruit to obligate vegetated nursery habitat, usually littoral seagrass beds that form a stable community, but also to algal beds and some seagrasses that may be ephemeral. They shelter and grow among the seagrasses until they reach about 12 mm CL (often smaller at about 9 mm for *P*. *semisulcatus*) when they begin to move into deeper waters (usually from November to April annually in the GoC) (Loneragan et al., 1994). As they grow to adults, they move offshore to depths of at least 20 to 30 m.

The optimal growth of juvenile Brown Tiger Prawns occurs at 30° C and a salinity of 25-35; though they grow well over a range of salinity, e.g. at 15 and 45. They also grow well at 25 and 35° C (O'Brien, 1994). Juvenile Brown Tiger Prawns use seagrasses in different ways, depending on the morphology of the seagrass and the size of the prawn. However, all juveniles rely on the protection afforded by the seagrass leaf-structure to avoid predators (Kenyon et al., 1995; Haywood et al., 1998). Very small juveniles (< 4 mm CL) do not bury and rely on the structure provided by the seagrass for protection (Kenyon et al., 1995). Small juvenile Brown prawns (about 5-9 mm CL) often do not bury when among large-leaved seagrasses, while large juveniles (> 10 mm CL) bury in the substrates for differing proportions of the day, depending on the leaf-size of the seagrasses (Hill and Wassenberg, 1993). Like Brown Tiger Prawns, juvenile Grooved Tiger Prawns prefer seagrass habitats over non-vegetated habitats, though their individual behaviour among seagrasses is not as well investigated. Large juveniles and sub-adults bury in sand substrate even if vegetated; rather than cling to seagrass leaves as refuge/camouflage (Hill and Wassenberg, 1993). In seagrass beds, juvenile Tiger Prawns have a striped green to green/brown colour pattern which provides camouflage among the seagrasses. Their survival in seagrass habitat is much greater than on bare substrates (Kenyon et al., 1995; Haywood et al., 1998).

In the GoC, postlarval Blue-tailed endeavour prawns settle from the plankton, usually to seagrass beds or algal beds, from October to about January (Staples et al., 1985). They shelter and grow among the seagrasses until they move off into deeper waters (usually from November to April). The juvenile distribution, behaviour and offshore migration of Blue-tailed endeavour prawns have not been well studied. Their appearance in the offshore fishery from October to June (at <20 mm CL) suggests that they do not spend a long time in nursery habitats, but move off at a small size. They move to deeper waters as they grow and as adults can be found to depths of about 50 to 60 m. Most of the commercial catch is taken in depths between 30 and 40 m (Kenyon et al., 2011).

Rigorous studies of the distribution and abundance of juvenile Red-tailed endeavour prawns in the NPF have not been undertaken. Postlarval Red-tailed endeavour prawns are found on most of the habitats that are available in estuaries, including seagrass, mangroves and channels (Staples et al. 1985). They do not favour any one habitat type. They shelter and grow in littoral habitat until they move off into deeper waters (usually from January to June annually in the western GoC). They move to deeper water and as adults are found to depths of 95 m (in the Joseph Bonaparte Gulf). Most of the commercial catch is taken in depths between 30 and 50 m and they are common in the north-west GoC (Kenyon et al., 2011).

Brown and Grooved Tiger Prawns are sympatric as adults. However, each species is consistently more abundant in some regions and at some times of the year (Kenyon et al., 2011). Their distribution is closely related to substrate type; Brown Tiger Prawns favour sandy sediments, whereas Grooved Tiger Prawns prefer sediments with a greater portion of mud (≥75%) (Somers, 1987). In the GoC, Grooved Tiger Prawns are abundant north of Groote Eylandt from July to November, whereas Brown Tiger Prawns are abundant in the vicinity of Mornington Island from May to August.

In the NPF, Blue- and Red-tailed endeavour prawns are sympatric as adults. However, each species is consistently more abundant in some regions and at some times of the year. Their distribution is closely related to substrate type; Blue-tailed endeavour prawns are found on sandy sediments (<50% mud), whereas Red-tailed endeavour prawns favour sediments with a greater portion of mud (>60% mud) (Somers, 1987).

As adults, both Tiger and Endeavour prawns bury during the day and emerge to feed at night (when they are fished). The diet of Tiger Prawns consists of small bivalves, gastropods, ophiuroids, crustaceans and polychaete worms (Wassenberg and Hill, 1987). Bivalves and gastropods are the most common food of juvenile and adult Brown and Grooved Tiger Prawns, and crustaceans are also common in the diet of Grooved Tiger Prawns. Endeavour prawns are carnivorous benthic feeders. Their diet consists of small molluscs, crustaceans, polychaete worms and Foraminifera. In turn, both Tiger and Endeavour prawns are preyed upon by fish (Sharks and teleosts), squid and cuttlefish (Brewer et al., 1991; 1995).

The interruption of first-season floods and the reduction of both low and high floodflows by water diversion or impoundment will probably not have negative effects on the growth, emigration of mortality of juvenile Tiger or Endeavour prawns. Tiger Prawns grow well in marine waters, and growth does not improve in brackish conditions. They experience ontogenetic cues for emigration from juvenile habitats to offshore zones. Turbid waters associated with floodflows may negatively impact their coastal habitats due to turbid coastal waters limiting light penetration to seagrass communities, and possibly sedimentation. However, following short-term negative impacts, the transfer of nutrients to the coastal zone via floodplumes may invigorate coastal habitats in the medium term.

Risk score justification

The four penaeid species complete the majority of their life cycle outside of estuaries (including offshore spawning) and have very little reliance upon estuaries to complete their life cycle (Figure 2.27). Although there is some use of mangrove areas by a proportion of advected juveniles, a reduction in flow is not likely to impact these species at the population level. In fact, it is possible these species may benefit from reduced flows that would decrease turbidity and potentially increase the amount of preferred seagrass habitat.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

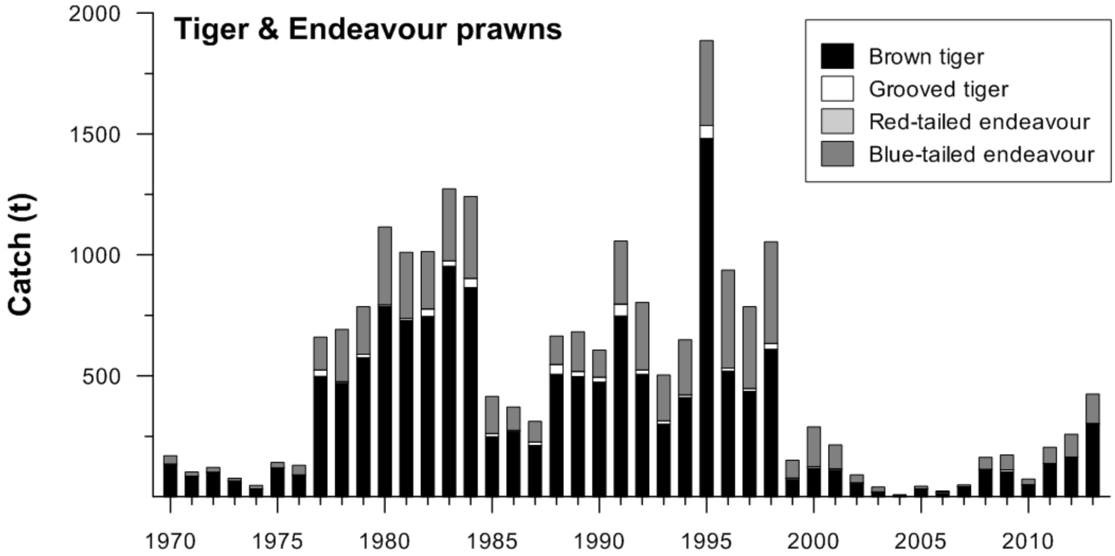

Figure 2.26. Annual catch of Grooved and Brown Tiger Prawns (*Penaeus semisulcatus* and *P. esculentus*) and Blue-tailed and Red-tailed endeavour prawns (*Metapenaeus endeavouri* and *M. ensis*) by the Northern Prawn Fishery in reporting Zone 8 in the Gulf of Carpentaria. Data supplied by the CSIRO.

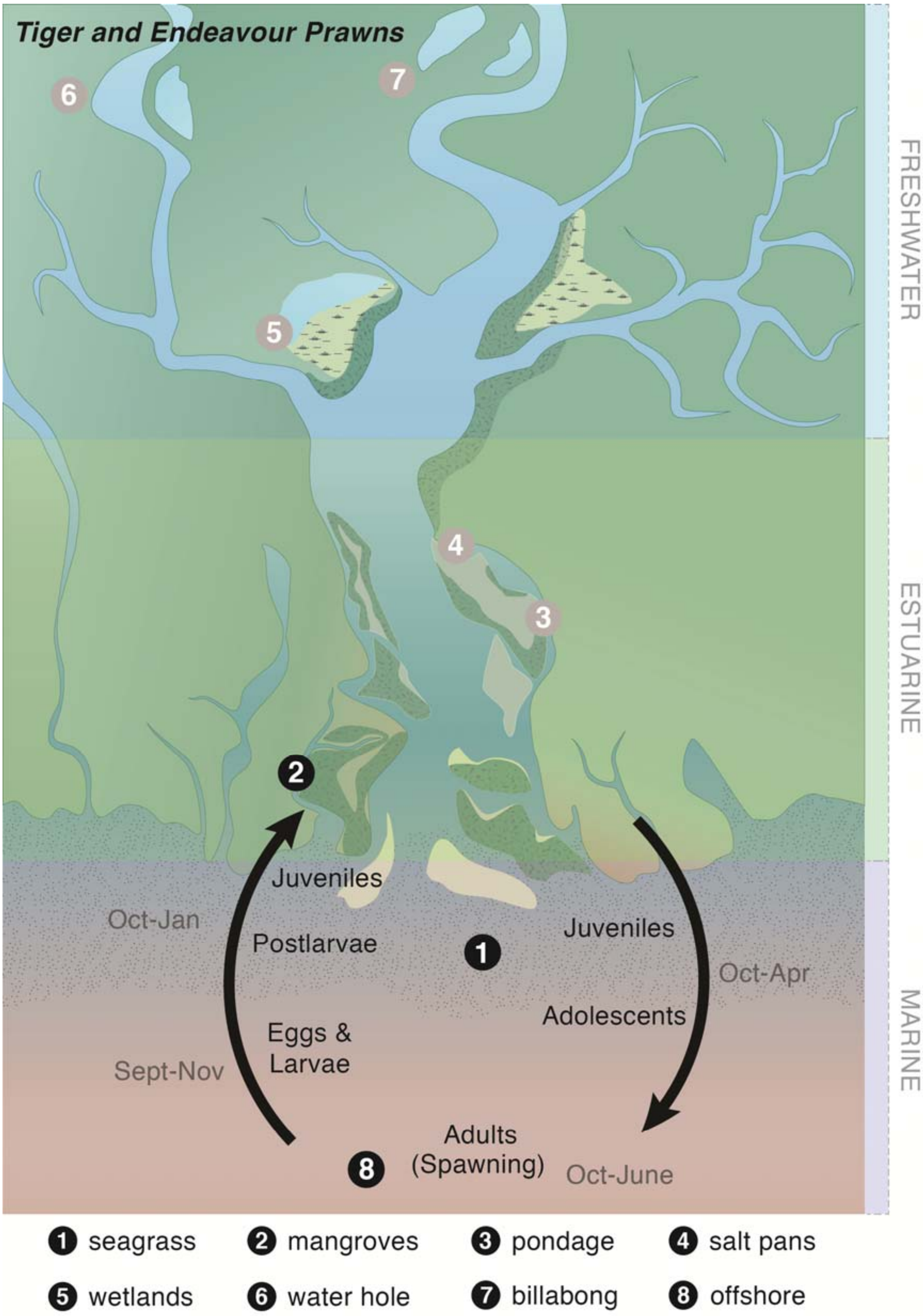

Figure 2.27. Conceptual model of the life history of Grooved and Brown Tiger Prawns (*Penaeus semisulcatus* and *P. esculentus*) and Blue-tailed and Red-tailed endeavour prawns (*Metapenaeus endeavouri* and *M. ensis*), illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.5 Mud Crab (*Scylla serrata*)

The Mud Crab ,*Scylla serrata*, supports a major inshore GoC fishery and typifies a life history strategy that would be significantly impacted by interruptions to the natural flows of the Flinders and Gilbert rivers. Of the two species of Mud Crab found in the GoC, *S. serrata* and *S. oliveria*, the former species is more abundant and supports fisheries in the region, and so it is the focus of this review.

The Mud Crab in the GoC has a larval life history strategy: the adults mate in the estuary and the females migrate to spawn in the offshore zone (September to November, larvae require marine salinity) (Hill, 1975; 1994). The eggs hatch to larvae that transform to megalopae that are advected inshore where they settle as benthic juveniles in estuarine mangrove and mudflat habitats (Meynecke et al. 2010). The larval form facilitates not only ontogenetic migration to their inshore habitats, but long distance dispersal and genetic mixing (Gopurenko and Hughes 2002, Gopurenko et al. 2004). They remain in the estuary for several years before the females alone emigrate to spawn offshore (Hill 1994). They are fished all year round. For some river systems, the annual wet season and subsequent runoff is a significant determinant of their recruitment strength and total catch (possibly lagged by one to two years) in the estuary and nearshore (Meynecke et al., 2010).

Mud Crab have been fished in the GoC since the 1970s and over 40 years of scientific research has explored the behaviour, habitats, feeding, ontogenetic and movement upon each stage of their life history (Hill, 1980; Hyland et al., 1984; Heasman et al., 1985; Hill, 1994; Gopurenko and Hughes, 2002; Moser et al., 2003). Crabs across large areas of the tropics are genetically mixed stocks, but significant barriers to hydrologic connectivity (e.g. Cape York/Torres Strait) also create distinct populations (Gopurenko and Hughes, 2002).

Mud Crabs demonstrate a larval life history strategy where the megalopae, juveniles and adults use different estuarine and coastal habitats. However, it does not demonstrate ontogenetic emigration from its juvenile habitat. Both juvenile and adult habitats are associated with well-established mangrove forests (Meynecke et al., 2010). The adults' habitats are spatially differentiated sub-tidal zones of the estuary or the close-in coastal zone (Hill et al., 1982; Hyland et al., 1984). Juvenile crabs (especially 20 to 80 mm CW) are resident in the intertidal zone where they select for structured habitat (Webley et al., 2009); whereas subadult (100 to 150 mm CW) and adult crabs (> 150 mm CL) move into the intertidal zone on the flood tide, but retreat to the sub-tidal zone as the tide recedes. Adult males remain in the littoral zone, while ovigerous females emigrate offshore to spawn.

The relationship between environmental cues and Mud Crab catch is not as clear as for some fishery species. In southern Australia (New South Wales), temperature explained 30 to 50% of the seasonal and monthly Mud Crab catch. However, in northern Australia there is a correlation (30 to 40%) between the Southern Oscillation Index (SOI) and subsequent fishery catch (e.g. the Flinders River) (Meynecke et al., 2010). High positive SOI is associated with higher rainfall and temperatures in coastal Queensland, so the correlation between crab catch and environmental flow (runoff lagged by six months or one year) from south-east GoC rivers is supported (Meynecke et al., 2010). Positive correlations between flow and crab catch have been identified in southeast and central Queensland previously (Loneragan and Bunn, 1999; Robins et al., 2005), though flow seasonality and two-year-lag effects suggest that relationship is not clear-cut. In the GoC, large floods such as the 1-in-50 year flood of 2009 and subsequent large floods can reduce the coastal catch of Mud Crabs, as the inshore zone becomes a freshwater habitat and crabs migrate elsewhere, due to their intolerance of very low salinities (Gary Ward, GoC fisher, pers. comms.).

Juvenile Mud Crabs that are resident in estuaries also benefit from low floodflows that create brackish conditions in the estuary. Though larvae survive best in marine salinity waters, the growth and mortality of juvenile Mud Crabs is optimal in waters characteristic of the tropics that are brackish: about 30 to 32°C and a salinity of 10 to 20 (growth) and 12 to 20 (survival) (Ruscoe et al., 2004; Meynecke et al., 2010). Estuaries in the southern GoC become hypersaline in the lead up to the wet season and in years of very low rainfall.

Under hypersaline conditions, growth and survival of the crabs may be inhibited until first rains and low flows reduce the salinity in the tropical estuaries to brackish levels.

Mud Crab is a highly important species in the GoC for commercial, recreational and Indigenous fisheries. The species is the primary target species of the Queensland pot fishery, but is also an important byproduct species in the N3 fishery. The total commercial catch of Mud Crab in the GoC has steadily increased from 23 t in 1993 to 199 t in 2012. Catches have remained reasonably stable over the past five years averaging 183 t per year (Figure 2.28).

Although the charter catch averaged only 682 kg in the GoC over the past five years, the 2010 statewide survey estimated 13,000 Mud Crabs were caught (Table 2.5), equating to around 13 t assuming an average weight of 1 kg. However, approximately 50% of this catch was released (QFISH online database). Similarly, the 2000/2001 NRIFS (Henry and Lyle, 2003) estimated 12,874 Mud Crabs were caught by Indigenous fishers, equating to 12.8 t. However, it’s important to know that this estimate is for all of Queensland.

Risk score justification
The Mud Crab relies strongly on estuaries to complete its life cycle (Figure 2.29). The interruption of first-season floods by in-stream dams and the reduction of both low and high floodflows by water diversion or impoundment may have a negative effect on the growth and mortality of Mud Crabs. In particular, the growth and survival of juveniles in the summer may be inhibited if first-season floods are impounded and estuaries remain hypersaline. There is some evidence of population subdivision at the estuary level, and consequently, a reduction in flow may reduce the number of recruits to the spawning population and affect the population size in the region.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

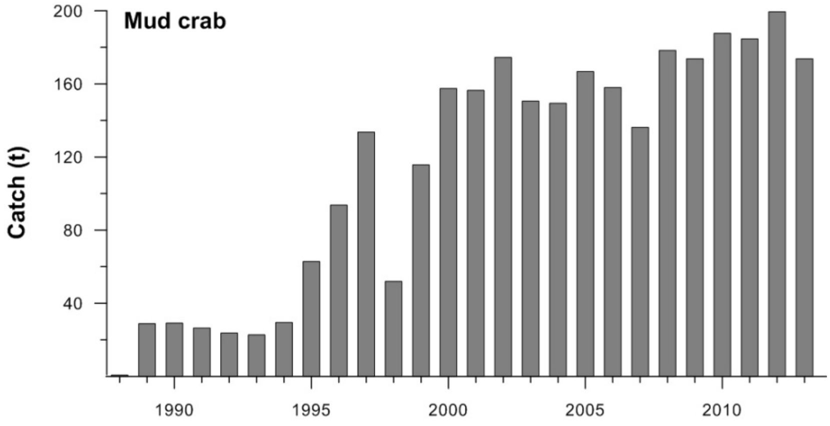

Figure 2.28. Annual catch of Mud Crab (*Scylla serrata*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

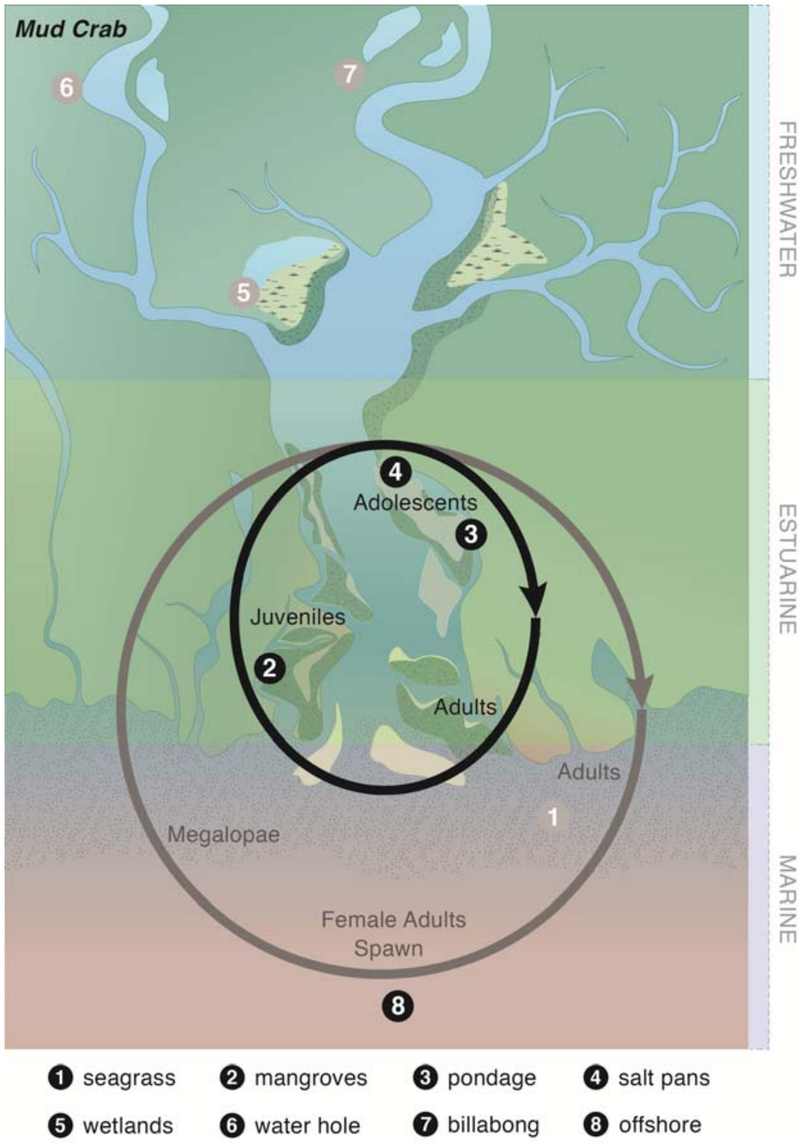

Figure 2.29. Conceptual model of the life history of Mud Crab (*Scylla serrata*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.6 Sawfish and Freshwater Whipray (*Himantura dalyensis*)

Sawfish and Freshwater Whiprays are EPBC-listed and IUCN-listed species that are threatened globally. Though they have or had pan-tropical or Indo-Pacific distributions, sawfish are rare or geographically extinct over large parts of their former range. Tropical Australia probably represents the last secure populations of Green Sawfish and dwarf sawfish (Pillans, 2014). Distinct population structures means Australia probably represents the last secure population of largetooth sawfish in the Indo-West Pacific, and probably globally (Pillans 2014). The GoC is a stronghold for all sawfish species. The Joseph Bonaparte Gulf, the eastern GoC and the southern coast of Papua New Guinea are the known distribution of the Freshwater Whipray (Last and Stevens, 2009).

Largetooth sawfish (*Pristis pristis*) is an EPBC-listed (vulnerable) and IUCN-listed species (critically endangered) that is threatened and iconic and typifies a life history strategy that would be significantly impacted by loss of longstream connectivity in the Flinders and Gilbert rivers. Loss of connectivity would include both the placement of barriers to migration and the loss of baseflow and floodflow that supports longstream connectivity between waterholes in the river bed.

The Largetooth Sawfish uses freshwater and marine habitats in the GoC; from nearly the full extent of rivers such as the Gilbert and Flinders to 100 km or so offshore (Pillans, 2014). It has an ontogenetic shift in habitat preference with juveniles using freshwater reaches of rivers and upper estuaries, and adults using lower estuarine and marine environments. Sawfish are born near the mouths of estuaries or nearshore coastal zone and migrate upstream to spend their first years of life in all-available freshwater reaches of the rivers and tributaries (up to 400 km inland) (Thorburn et al., 2004; Pillans, 2014). As they mature (240 cm – males; 280 cm – females), Largetooth sawfish move out of the river and enter the coastal marine environment. As adults (up to 600 cm), the species can be found in the lower marine-influenced estuary, and nearshore and offshore marine zones (Pillans, 2014). Sawfish are caught as incidental catch in both the NPF trawl fishery and inshore gillnet fisheries (Barwick, 2013). In 2011, a total of 12 Largetooth sawfish were recorded in the N3 inshore gillnet fishery (Queensland, 2014).

Although aspects of its reproduction is a significant knowledge gap, Largetooth sawfish likely enter less saline waters to give birth, possibly in the late wet season. The capacity of juveniles to recruit to their riverine freshwater habitats may be impacted by wet-season water levels in Australian tropical rivers.

The Green Sawfish (*Pristis zijsron*, listed as vulnerable under EPBC) has a similar life history to Largetooth sawfish, except that it does not use the freshwater reaches of rivers. Its juvenile phase likely uses estuaries and bays, including brackish reaches. As adults (~500 cm) although they are caught offshore, they tend to remain in the inshore coastal fringe associated with mangroves and adjacent mudflats. The Green Sawfish likely give birth just before or during the wet season in the Australian tropics.

The Dwarf sawfish (*Pristis clavata*, vulnerable EPBC) is endemic to Australia and it has a similar life history to Green Sawfish; it inhabits shallow (2 to 3 m) inshore and estuarine habitats. It is mostly found in marine waters in the lower estuary, though is caught in brackish water (salinity 9.7). Juveniles use estuaries as nursery habitat and shows site fidelity occupying a range of a few km^2 in the coastal fringe and tracked individuals have been shown to rest for ~100 minutes at close-by sites in inundated mangrove forests on consecutive high tides. The Dwarf Sawfish likely gives birth during the wet season in the Australian tropics.

The Freshwater Whipray (*Himantura dalyensis* (Last and Manjaji-Matsumoto, 2008), previously *Himantura chaophraya*) has been recorded from the Gilbert River (Last and Stevens, 2009). It has a life history similar to the Largetooth sawfish. Freshwater Whiprays are born in the estuary and migrate upstream to spend their first years of life in all-available freshwater reaches of the rivers and tributaries (> 300 km inland) (Thorburn et al., 2003; Burrows and Perna, 2006). As the Whiprays mature, they move down the river and enter the estuarine environment. As adults (up to 160-200 cm disc) they can be found in the upper marine-influenced estuary; they have not been recorded in euhaline marine waters (30 to 40) and most are found in waters < 10 (Thorburn et al., 2004). Though the Freshwater Whipray population is most likely sustainable, co-generic species in Asia are listed as endangered, and locally critically endangered.

Although much more needs to be understood about the reproduction, feeding and predation of sawfish, all species seem to pup during the wet season, possibly at the end of the wet season. They may prefer waters that are not fully saline in which to pup as the juveniles prefer brackish habitat (and freshwater habitat in the case of Largetooth sawfish). Inshore brackish habitats may support higher abundances of prey items for juvenile sawfish and the turbid waters of flooded inshore habitat may provide juveniles with protection from their predators. Turbid water may also allow sawfish to approach their prey within range of their rostrum. Stomachs of Largetooth sawfish contained penaeid prawns, eel tailed catfish, jewel fish, mullet, threadfin and freshwater prawns (Pillans, 2014). Specimens of Largetooth sawfish collected for aquaria from the GoC have had the scales of quite large fish prey impaled on their rostrums: Barramundi, Saratoga and Jewfish. Whiprays are active predators of fish and prawns in estuaries (Last and Stevens, 2009). So both their prey species, and sawfish and whiprays themselves, are impacted by interruptions to natural flows (see reviews for other fish species).

Thus, the interruption of wet season low-flows by in-stream dams and the reduction of both low and high floodflows by water diversion or impoundment will affect the foraging and refuge habitats of all sawfish. Flow interruption may have a negative effect on reproduction of green and dwarf sawfish; and will have a negative impact on the ability of juvenile Largetooth sawfish and Freshwater Whiprays to access their freshwater habitats. Physical barriers to longstream connectivity such as in-stream dams, barrages and road bridges constructed for access to these newly-developed landscapes interrupt their movement to freshwater habitats and return to the estuary. These effects will impact negatively on the survival of these critically endangered species.

Risk score justification

Sawfishes and the Freshwater Whipray spend a significant proportion of their life cycle within estuaries (Figure 2.30). Largetooth sawfish and Freshwater Whipray in particular require access to freshwater habitats, which would be severely impeded by river development upstream. These species also have slow growth, are long lived, and there is strong evidence for population subdivision at the estuary level. Consequently, a reduction in flow is likely to significantly reduce the population size and dynamics in the GoC. Flow effects would however, be less important to populations of Green and Dwarf sawfish species.

Largetooth sawfish and Freshwater Whipray

Risk scores: Consequence **4**; Likelihood **5**. **Overall risk rating**: HIGH

Green and Dwarf sawfish

Risk scores: Consequence **3**; Likelihood **3**. **Overall risk rating**: MODERATE

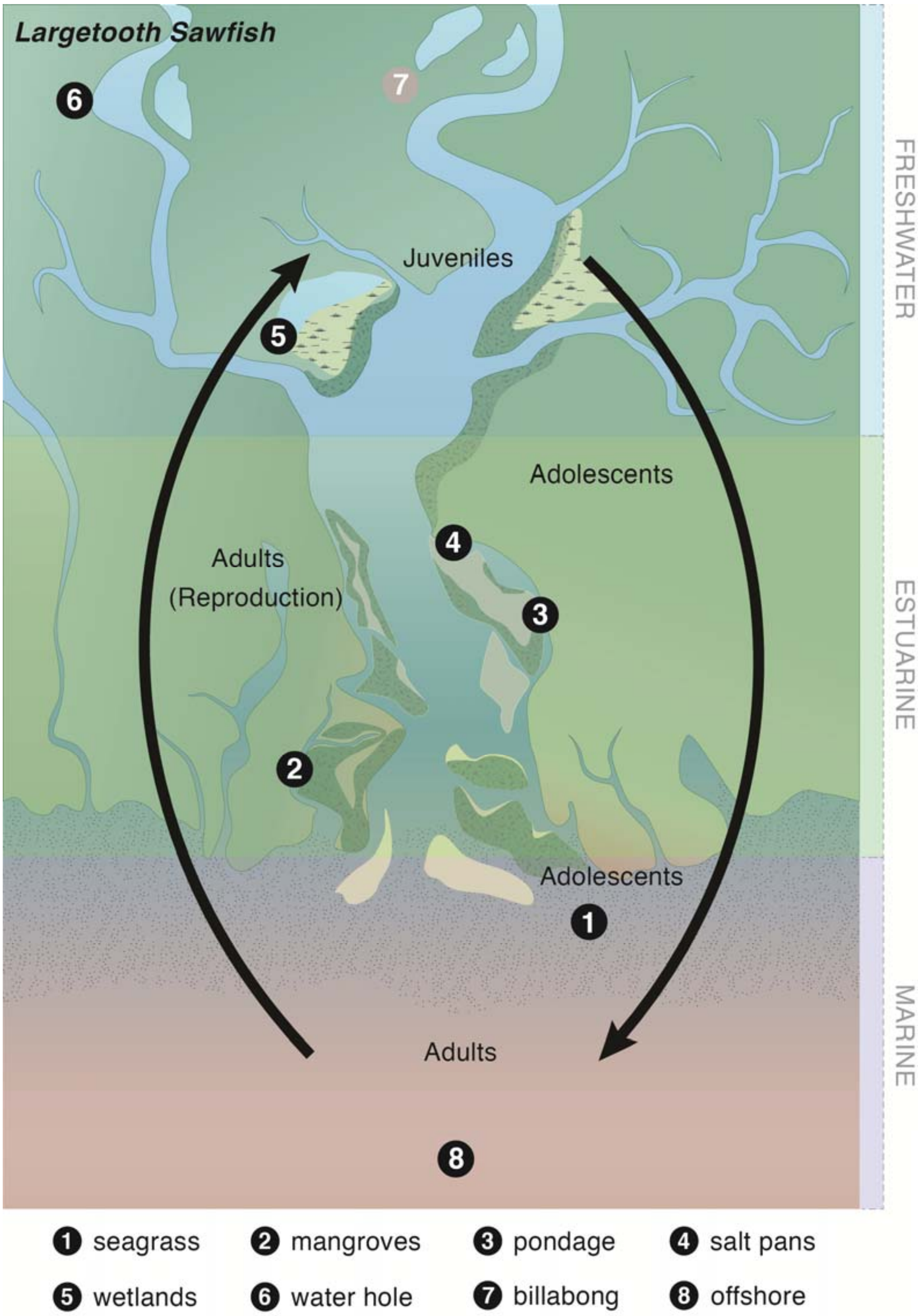

Figure 2.30. Conceptual model of the life history of Largetooth sawfish (*Pristis pristis*) and Freshwater Whipray (*Himantura dalyensis*) illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.7 River Sharks (*Glyphis glyphis* and *G. garricki*)

River Sharks is the generic term given to species of the genus *Glyphis*, within which there are two species found in Australian waters, the Speartooth Shark (*Glyphis glyphis*) and the Northern River Shark (*G. garricki*). The Speartooth Shark is endemic to Cape York, the west coast of the Top End and the southern coast of Papua New Guinea. The Northern River Shark is endemic to the Kimberley/Top End coast and the Fly River, Papua New Guinea. Tropical Australia probably represents the last viable populations of Speartooth Shark and Northern River Shark across their global ranges (Pillans, 2014). Freshwater Sharks are of significance to the Indigenous people of the region. Comments from elders seem to reflect a seasonal appearance of sharks in the freshwater reaches of the rivers, which along with other species were food sources. "The different seasons are there for different things. Just before the wet, it is the sharks, then the big white catfish and the Cherabin come in the wet, and we collect heaps" (Barber, 2013).

Speartooth Shark (*Glyphis glyphis*) is an EPBC and IUCN-listed species (critically endangered) that is threatened and iconic and typifies a life history strategy that would be significantly impacted by loss of longstream connectivity of GoC rivers. In Queensland, the species is found on western Cape York in Port Musgrave, the Wenlock and Ducie Rivers. The western Cape rivers further south were likely their former (and possibly current) range. This species was more widespread in the recent past and may use other GoC rivers (Pillans, 2014). Although neither species of rivershark has not been recorded from the Flinders and Gilbert rivers, these rivers provide suitable habitat. Therefore, with surveys undertaken at favourable times and habitats may determine the presence of absence of this species in these rivers. Loss of connectivity would include both the placement of barriers to migration and the loss of baseflow and floodflow that supports longstream connectivity between waterholes in the river bed.

The Speartooth Shark uses large tropical river systems as its primary habitat (salinity 0.8-28) (Pillans, 2014). In the GoC, it uses the lower extent of freshwater and the estuarine habitats of Gulf rivers. It has an ontogenetic shift in habitat preference with juveniles using the upper-estuarine and lower-freshwater reaches of rivers (up to 100 km upstream), and adults using estuarine environments (Pillans et al., 2009). All aspects of the reproduction of the Speartooth Sharks are significant information gaps; however, it is likely they are born from October to December in the lower estuary or near the mouths of rivers. They spend their early juvenile phase in estuarine and/or freshwater reaches of tropical rivers. Mature Sharks prefer highly turbid, tidal waters over fine muddy sediments. They move up and down the estuary with the flood and ebb tides (Pillans et al., 2009). Adults (likely > 200 cm) are found in the lower marine-influenced estuary; none have been recorded outside rivers in marine zones (Pillans et al., 2009; Field et al., 2013).

The Northern River Shark (*Glyphis garricki*) (EPBC and IUCN-listed species, endangered) uses freshwater habitats as well; yet it differs from the Speartooth Shark as it is more marine in habit. It uses rivers (salinity 2), large tropical estuarine systems (salinity 7–21), macrotidal embayments and inshore and offshore marine habitats (salinity 32–36) (Pillans et al., 2009). It is thought adults use only marine environments and may be found well outside estuaries. The Northern River Shark likely pups prior to the annual wet season with a litter size around nine. Neonates and juveniles are found in freshwater, estuarine and marine habitats, though capture locations indicate a preference for highly turbid, tidally influenced waters over muddy substrate (Stevens et al., 2005). No *Glyphis* species have been found in isolated freshwater habitats such as billabongs or refuge waterholes in river channels (Stevens et al., 2005).

Juvenile Speartooth Shark consume estuarine and freshwater fish and crustaceans including catfish, nurseryfish, Bony Bream, freshwater gobies and *Macrobrachium* spp. (Pillans, 2014). Pillans (2014) in his recent review revealed that the diet of adult Speartooth Shark is unknown. The Northern River Shark feeds primarily on bony fishes and stomach contents have contained King Threadfin, Fork-tailed Catfish and Barramundi, and they also seem to consume rays (Pillans, 2014). The preference of these Sharks for highly turbid waters may in part reflect a foraging strategy in turbid waters where fish and crustacean prey are easier to catch.

Risk score justification

The interruption of wet season low flows by in-stream dams and the reduction of both low and high floodflows by water diversion or impoundment are likely to have a negative impact on the ability of juvenile Speartooth Sharks to access freshwater habitats (Figure 2.31). Physical barriers to longstream connectivity such as instream dams and barrages may interrupt their movement to freshwater habitats and their return to the estuary (Figure 2.31). Barriers and flow reduction may also impact the movement of Northern River Shark in the upper estuary. The reduction of both low and high floodflows may have a negative impact on prey availability for these river sharks as some of the prey may also be impacted by flow reduction. These species also have slow growth, are long lived, and there is strong evidence for population subdivision at the estuary level for these endangered species. Consequently, a reduction in flow is likely to significantly reduce the population size and dynamics of the species in the GoC.

Risk scores: Consequence **4**; Likelihood **4**. **Overall risk rating**: HIGH

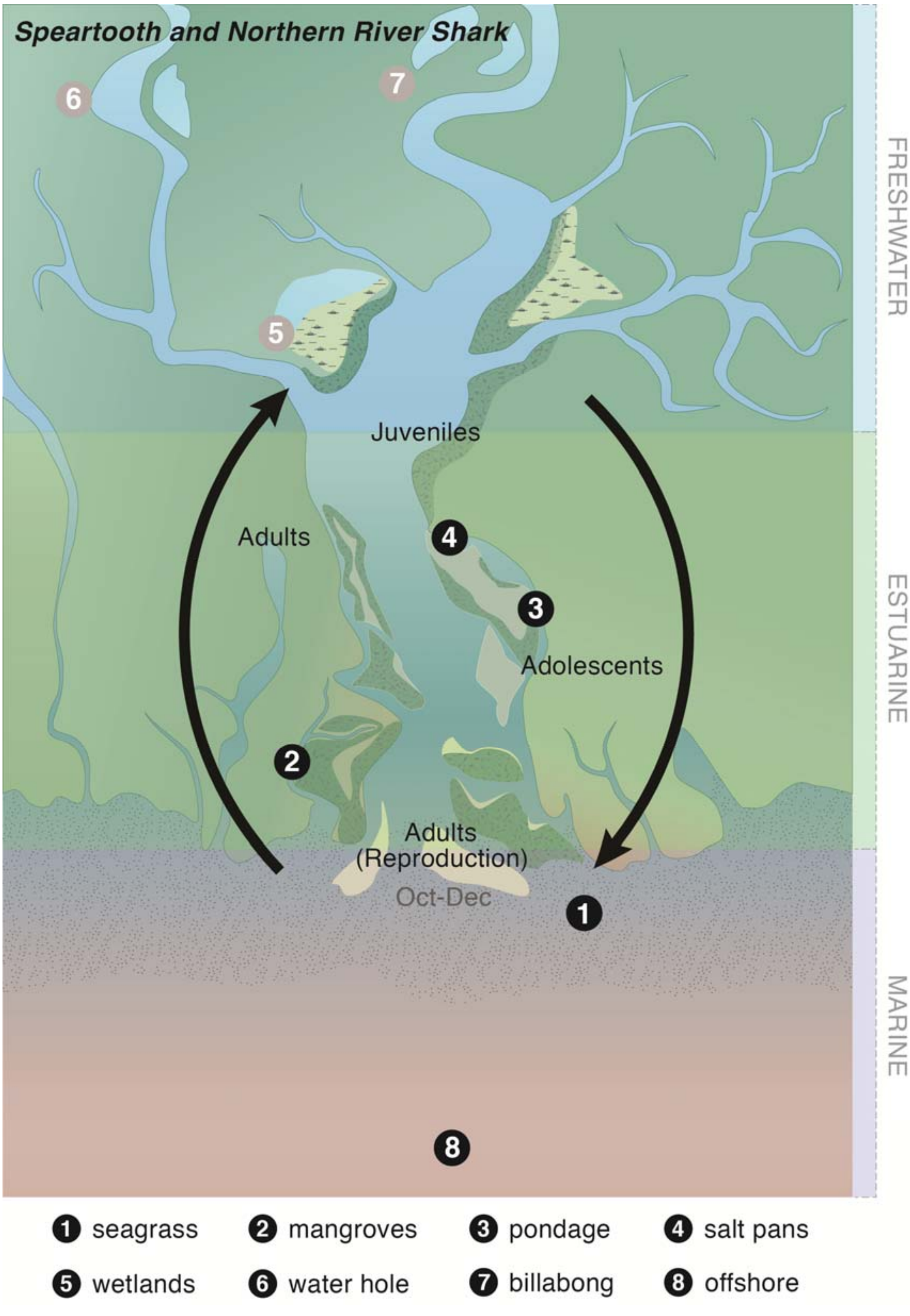

Figure 2.31. Conceptual model of the life history of River Sharks (*Glyphis glyphis* and *G. garricki*) illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.8 Australian Black-tip Shark (*Carcharhinus tilstoni*)

The Australian Black-tip Shark (*Carcharhinus. tilstoni*) is endemic to northern Australia, whereas related species occur more widely. *C. tilstoni* co-occurs with the common Black-tip Shark (*C. limbatus*) in Australian waters and the two are difficult to distinguish (Tillett et al., 2012) with hybridisation between the two species recorded (Morgan et al., 2012). Black-tips are a major component of commercial shark catches in northern Australia. *C. tilstoni* is distributed throughout continental shelf waters. However, all life stages are found in nearshore waters (Stevens et al., 2000b; Harry et al., 2011b). The Black-tip Shark feed primarily on fish (Salini et al., 1992; Farmer and Wilson, 2011) but are also significant predators of crustaceans such as penaeid prawns (Salini et al., 1992).

The Australian Black-tip gives birth to live young, around December and January (Robins and Ye, 2007; Harry et al., 2013). The juveniles are not strongly reliant on nearshore areas, and generally use them opportunistically (Robins and Ye, 2007). In Cleveland Bay on the east coast of Australia, the highest numbers of juvenile Sharks tend to occur when prey abundance is highest (Simpfendorfer and Milward, 1993a).

Carcharhinid sharks are the fourth most important species (by weight) in GoC commercial fisheries. A number of problems with identifying common species and logbook reporting by commercial fishers makes species-specific assessment very difficult. Prior to 2005, the Australian blacktip was a common species recorded as ‘Unspecified shark’, where catches ranged between 200 t and 500 t (Figure 2.32). Scientific observer data indicate that the ‘Black-tip whaler complex’ – comprising *C. tilstoni* and *C. limbatus* – is the most commonly caught Shark in the N3 and N12/13 fisheries. The improved logbook reporting from 2006 shows the Australian Black-tip catch has since ranged between 69 t and 130 t (Figure 2.32). Although catch data from Queensland fisheries are not available by fishery, most of the commercial catch comes from the commercial N12 and N13 offshore gillnet fishery, with substantial catches also from the N3 inshore gillnet fishery. It is not an important species in the charter or recreational fisheries, but may be reported in aggregate species groups such as ‘Unspecified Shark’ or ‘Whaler Sharks’.

Risk score justification

This species has a wide habitat envelope and generally prefers less turbid waters than estuaries (Figure 2.33). Therefore, it has little specific reliance on estuaries to complete its life cycle. The species may be affected by changes in flow through modification of prey availability, particularly for juvenile Sharks which are more prevalent in inshore regions (Fig 1: F1) (Robins and Ye, 2007), but a reduction in river flow of the Flinders and Gilbert rivers is unlikely to result in any detectable changes in population size or dynamics.

Risk scores: Consequence **0**; Likelihood **2**. **Overall risk rating**: NEGLIGIBLE

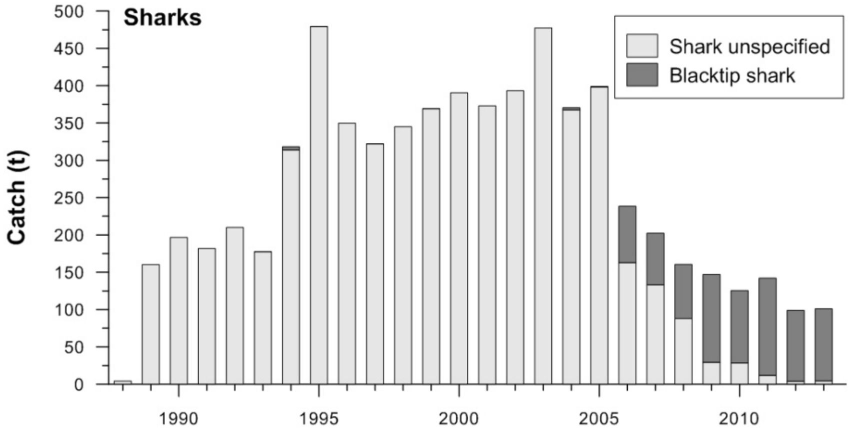

Figure 2.32. Annual catches of Australian Black-tip Shark (*Carcharhinus tilstoni*) and "Shark unspecified" by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

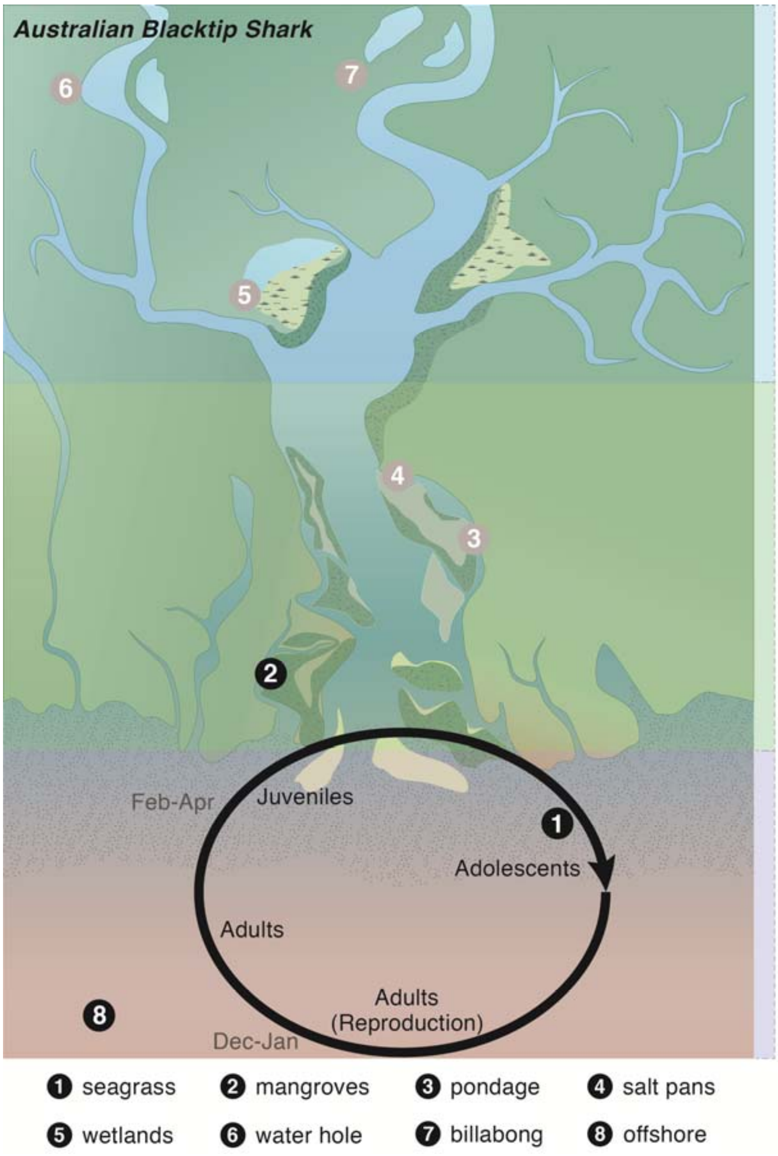

Figure 2.33. Conceptual model of the life history of Australian Black-tip Shark (*Carcharhinus tilstoni*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.9 Bull Shark (*Carcharhinus leucas*)

The Bull Shark (*Carcharhinus leucas*) is distributed throughout the tropical and warm temperate regions of the world (Last and Stevens, 2009). It prefers neritic waters, generally of less than 30 m depth, but has been found at depths of up to 150 m (Compagno, 1984). It is commonly found in GoC estuaries (Blaber et al., 1989; 1990; 1995; Salini et al., 1998), and has been documented travelling long distances upstream into freshwater (Thorson, 1971). It has an extremely wide salinity tolerance, even being observed in hypersaline conditions of up to 53 PSU.

The Bull Shark grows to a maximum length of 340 cm TL (Last and Stevens, 2009). A number of age and growth studies on the Bull Shark use a variety of methods from vertebral counts to tagging. The more reliable vertebral ageing methods of Branstetter and Stiles (1987) showed that the Bull Sharks is slow growing and long lived, estimating the oldest male and female fish to be 21.3 and 24.2 years, respectively. Smith et al. (1998) estimated a maximum age of 27 years.

Few studies have examined the reproductive dynamics of the Bull Shark. The species is gonochoristic with a male:female sex ratio of approximately 1:1. Branstetter and Stiles (1987) estimated that maturity of males and females occur very late in life at 14 to 15 years and 18+ years, respectively. Females are viviparous and give birth to 1 to 13 live young often in estuaries or freshwater rivers during late spring and summer (Last and Stevens, 2009). Juveniles then use estuarine habitats for up to several years before moving to deeper waters in the coastal and offshore zones, although adults often return to estuaries to feed (Snelson et al., 1984). In the Norman River, Salini et al. (1998) found the Bull Shark to be the third most important species in terms of biomass in fishery independent gillnet surveys. This study also found Bull Sharks to be one of the most important predators in the Norman River, consuming a variety of fish species as well as commercially important penaeids.

The population structure of the Bull Shark is little studied, but given its cosmopolitan distribution, it is highly likely that it exists as a single stock at least within the GoC, and most likely more widely.

The Bull Shark is not an important species to commercial fisheries in the GoC, but is retained as a byproduct species. Although catch data from Queensland fisheries are not available by fishery, most of the commercial catch probably comes from the commercial N3 gillnet fishery. Commercial catch data are only present for the years 2008 to 2010, which show a maximum catch of 4.5 t in 2009 (Figure 2.34). As with many species of carcharhinid Sharks caught in the commercial fishery, the level of reporting is often at the generic level of 'Whaler Sharks' or 'Unspecified Shark', and so Bull Shark is most likely included in these categories. The Bull Shark is not a target of the recreational fishery but small individuals are likely to be captured, particularly within estuaries.

Risk score justification

The Bull Shark uses estuarine and freshwater habitats throughout its entire life cycle (Figure 2.35), for breeding and feeding grounds as adults, and as a nursery as juveniles. The movement of this species would be significantly affected by reduced flows, thus reducing access to key habitats for feeding, breeding and pupping. Bull Sharks are slow growing, long lived and have low fecundity, indicating that a reduction in river flow of the Flinders and Gilbert rivers may result in a detectable change in population size or dynamics even at the wider scale of the GoC.

Risk scores: Consequence 3; Likelihood 4. **Overall risk rating**: HIGH

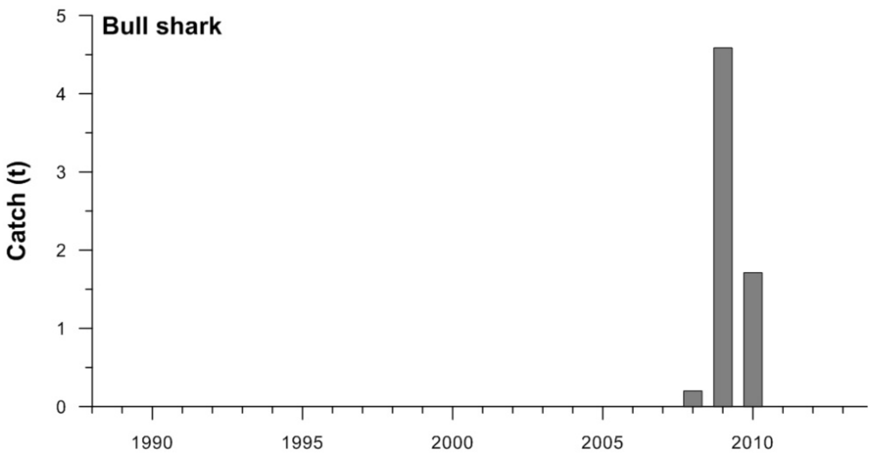

Figure 2.34. Annual catches of Bull Shark (*Carcharhinus leucas*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

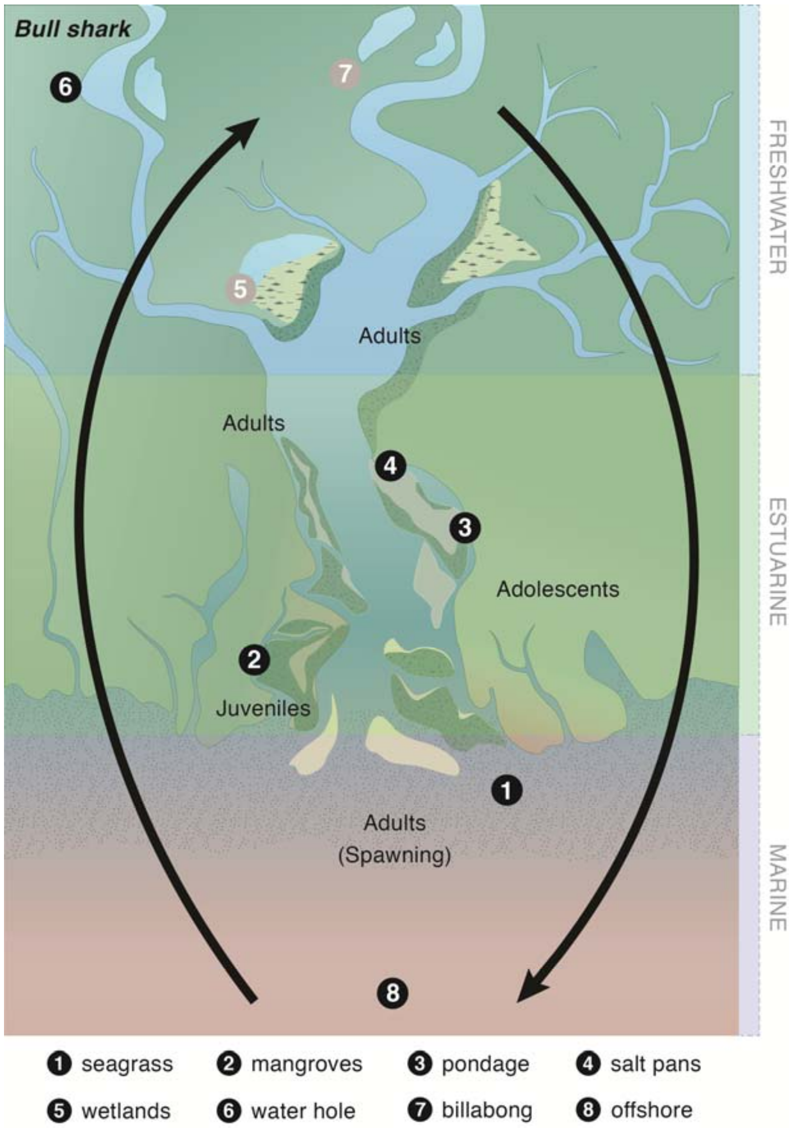

Figure 2.35. Conceptual model of the life history of Bull Shark (*Carcharhinus leucas*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.10 Hammerhead Sharks (Sphyrnidae spp.)

The GoC hosts three species of Hammerhead Sharks (family Sphyrnidae): the Great hammerhead (*Sphyrna mokarran*), Scalloped hammerhead (*Sphyrna lewini*) and Winghead Shark (*Eusphyra blochii*). All three species have been recorded in the catches of commercial net fisheries in the GoC. The former two species are usually found more commonly in offshore waters (Compagno, 1984), whereas the Winghead Shark tends to prefer more turbid coastal waters. Winghead Shark comprised most of the catch of Hammerheads in the GoC coastal commercial net fisheries (unpublished observer data, DAFF), as well as in fishery-independent surveys by the CSIRO in the Norman River (Salini et al., 1998). Winghead Shark will therefore be the focus of this review.

The Winghead Shark is distributed throughout the tropical and subtropical regions of the Indo-West Pacific and is common across northern Australia. It is an important predator in these habitats feeding mainly on fish and crustaceans, in particular, commercially important penaeids in the GoC (Salini et al., 1998).

Very little is known of the biology of the Winghead Shark. The species grows to a maximum size of 186 cm TL (Last and Stevens, 2009) and there has been only one published study on its growth, based on only 14 individuals from the Great Barrier Reef (Smart et al., 2013). Their study revealed that the Winghead Shark is slow growing and long lived, with the oldest fish aged at 21 years. These results agree with the life history of closely related species such as the Scalloped and Great hammerheads that have been reported to grow slowly and live for up to 30 years (Piercy et al., 2007; Harry et al., 2011a).

Limited information is available on the reproductive dynamics of the Winghead Shark. The species is gonochoristic with a reasonably even sex ratio, and females in northern Australia reach maturity at around 120 cm TL. Females breed every year and are viviparous, giving birth to around 12 live young between January and March after a gestation period of 10 to 11 months (Stevens and Lyle, 1989). It is not well understood what habitats juveniles use as nursery habitats, but it is likely they use shallow coastal embayments and the lower reaches of estuaries, which have been documented as nursery areas for Scalloped hammerheads (Blaber et al., 1989; Simpfendorfer and Milward, 1993b; Blaber et al., 1995)

The population structure and movements of Winghead Sharks is little known given its low importance to fisheries and the subsequent low priority for genetic studies and tagging programs to be established. Stevens et al. (2000a) tagged 34 Winghead Sharks across northern Australia but only recorded one tag return of an individual that was at liberty for one year, but unfortunately no data were provided as to the extent of movement. Given the wide distribution of this species, there is little evidence for population subdivision, as is the case with the co-occurring Scalloped hammerhead in Australia (Ovenden et al., 2011).

Hammerhead Sharks are reasonably important to inshore commercial gillnet fisheries, but are of little importance to recreational and Indigenous fisheries. Although catch data from Queensland fisheries are not available by fishery, most of the commercial catch comes from the commercial N3 gillnet fishery. There are significant issues with identification and logbook reporting of hammerhead Sharks, and so the available logbook data are of little use in investigating species-specific catch trends over time. The three species of hammerheads in the GoC have been reported generically as “Hammerhead Shark”, Scalloped, and Winghead hammerheads. The catch of Scalloped hammerheads as the only hammerhead species to be reported from 2004, which peaked at 78 t in 2006 and was not reported again after 2007. It appears that a change in logbook reporting occurred from 2007 where only Winghead and ‘Hammerhead Shark’ were reported (Figure 2.36). The Winghead Shark catch is reasonably low peaking at 8 t in 2010, but it is likely that this species is included in the ‘unspecified Hammherhead Shark’ catch. Although Hammerheads were recorded to be caught in the study region in the 2010 statewide recreational fishing survey, no catch estimate was made and is likely to be low.

Risk score justification

Scalloped and Great hammerhead Sharks are widely distributed and primarily occur and reproduce in offshore waters (Figure 2.37). These species are of conservation concern due to their low productivity, but it is unlikely that reduced flows in the Flinders and Gilbert rivers would have a significant impact on their entire populations. The Winghead Shark however, prefers more turbid nearshore habitats and there is evidence to suggest the species uses nearshore and lower estuarine habitats for feeding, and possibly as a nursery for juveniles. The species may be affected by changes in flow through modification of prey availability, particularly for juvenile Sharks. The species is slow-growing, long lived and has low fecundity, indicating that a reduction in river flow of the Flinders and Gilbert rivers may result in a detectable change in population size or dynamics.

Scalloped and Great hammerhead
Risk scores: Consequence 1; Likelihood 3. **Overall risk rating**: LOW

Winghead Shark
Risk scores: Consequence 2; Likelihood 3. **Overall risk rating**: MODERATE

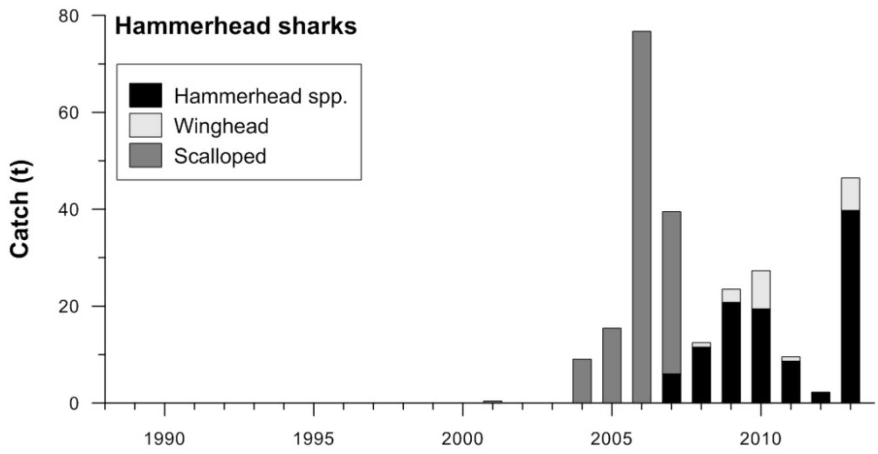

Figure 2.36. Annual catch of Hammerhead Sharks (family Sphyrnidae) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

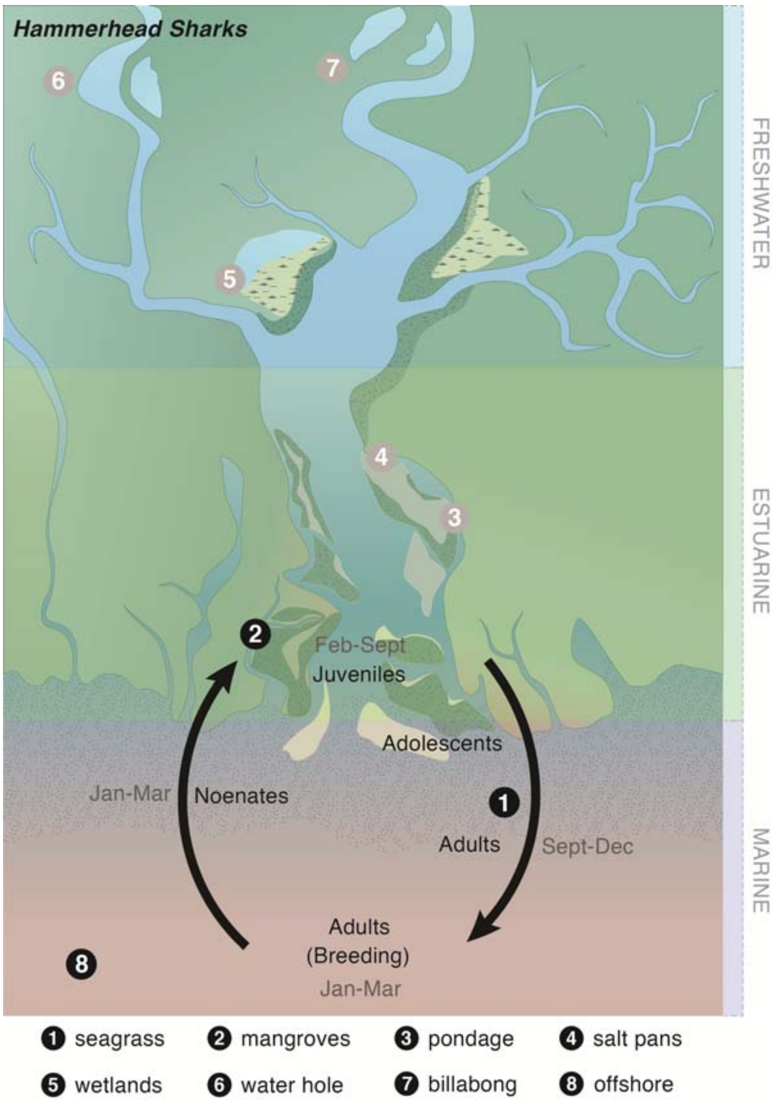

Figure 2.37. Conceptual model of the life history of Hammerhead Sharks (family Sphyrnidae), illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.11 Grey Mackerel (*Scomberomorus semifasciatus*)

Grey Mackerel (*Scomberomorus semifasciatus*) is a fast-growing pelagic species endemic to the waters of northern Australia and Papua New Guinea (Ward and Rogers, 2003; Welch et al., 2009; Juan-Jorda et al., 2013). Although the species can tolerate a wide range of salinities (9 to 36.0 PSU) and turbidity (0.8 to 11.0 NTU) in the GoC (Cyrus and Blaber, 1992), it generally prefers shallow marine waters around rocky headlands and inshore reefs (Ward and Rogers, 2003; Welch et al., 2009; 2014) and estuarine waters where both salinity and turbidity are reasonably high, where they feed on small pelagic fish (Salini et al., 1990; Salini et al., 1998).

In the GoC, there are major seasons for fisheries on Grey Mackerel related to seasonal aggregations, hence availability, of the fish (Welch et al., 2009). It has been suggested that freshwater flows into coastal waters, stimulating productivity, are an important cue for aggregating (Ward and Rogers, 2003; Robins and Ye, 2007).

Grey Mackerel spawn between August and December across northern Australia (Welch et al., 2009). However, spawning may occur earlier in the eastern GoC with the spawning season possibly starting as early as May (Welch et al., 2009). Although little is known of the specific spawning locations of Grey Mackerel, it is likely to spawn in offshore waters throughout the region as do other species of pelagic fish (Griffiths et al., 2006; Fry and Griffiths, 2010). Larvae and juveniles move into inshore and estuarine habitats (Jenkins et al., 1984; Halliday and Robins, 2005) for shelter and feeding, so growth and survival are likely to be influenced by river flows (Welch et al., 2014).

There has been a significant amount of research on the population structure of Grey Mackerel. Genetic, parasite and otolith microchemical analyses reveal there are at least five stocks of Grey Mackerel across northern Australia: Western Australia, north-west Northern Territory, GoC and the north east and central east coast (Welch et al., 2009; Newman et al., 2010a; Broderick et al., 2011).

Grey Mackerel is the second most important commercial species (by weight) in the GoC. Although catch data from Queensland fisheries are not available by fishery, most of the commercial catch comes from the commercial N12 and N13 offshore gillnet fishery, with substantial catches also coming from the N3 inshore gillnet fishery. It is not an important species to the charter or recreational fishery.

The commercial catch of Grey Mackerel in the GoC has steadily increased from 378 t in 1997 to a peak of 897 t in 2010 (Figure 2.38), with the average annual catch in the past five years being 767 t. Grey Mackerel has not been explicitly recorded in the charter fishery, but small numbers are likely to be caught and recorded as 'unspecified Mackerel' (DAFF unpublished logbook data). This species was not recorded as being captured by recreational fishers in the GoC in the 2010 statewide recreational fishing survey.

Risk score justification

Given that Grey Mackerel uses the lower parts of estuaries in its first few months of life (Figure 2.39), but also other sheltered habitats (e.g. bays), a change in river flows of the Flinders and Gilbert rivers may result in small changes in population size or dynamics that may be detectable. This may be due to a negative impact on larval and juvenile growth and survival through changes in prey availability or extent and connectivity of estuarine habitat. Further, reduction in wet season flow may reduce the turbidity of the offshore flood plume that adults may normally use to ambush prey, thus reducing their feeding success.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

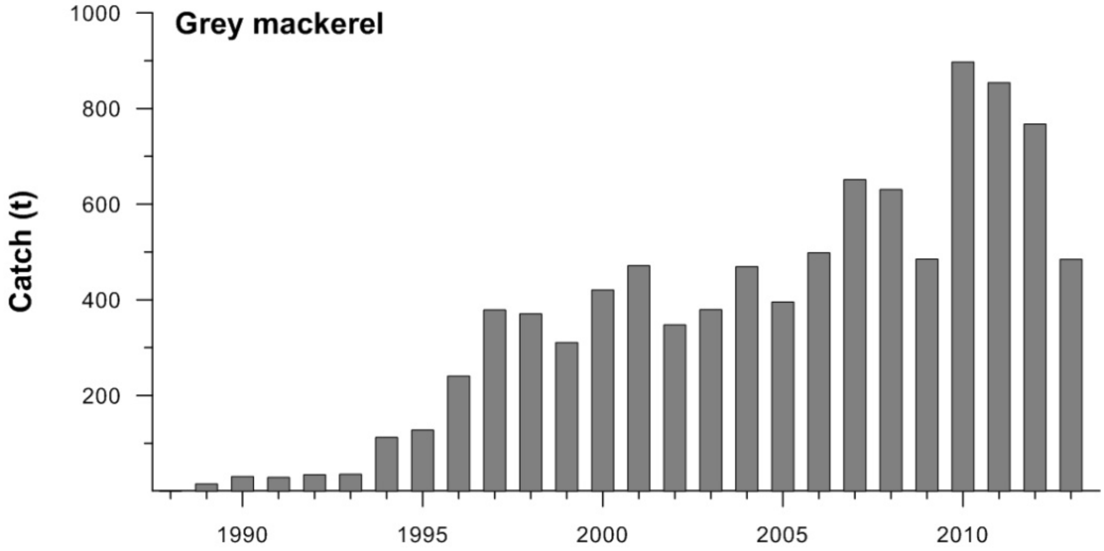

Figure 2.38. Annual catch of Grey Mackerel (*Scomberomorus semifasciatus*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

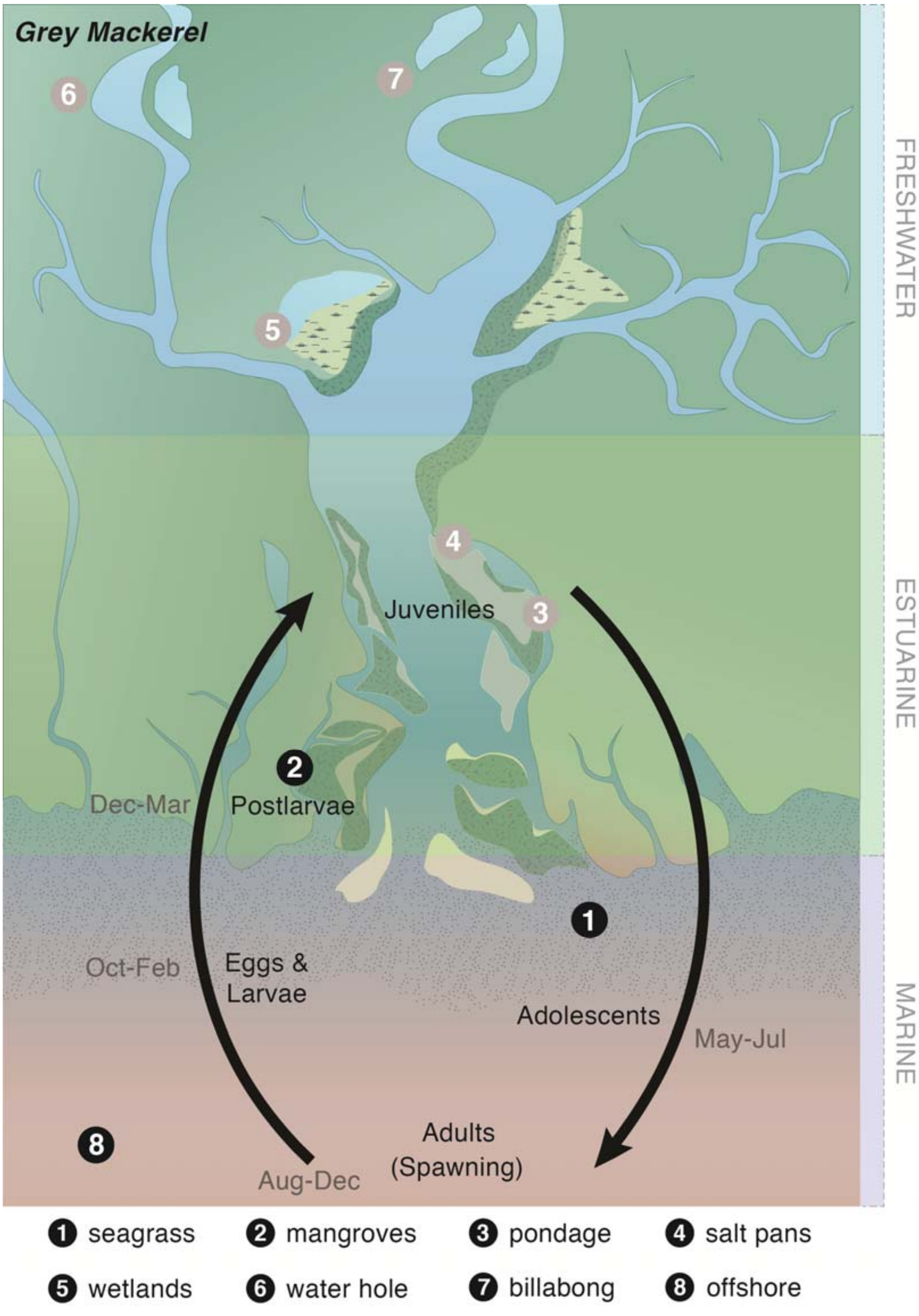

Figure 2.39. Conceptual model of the life history of Grey Mackerel (*Scomberomorus semifasciatus*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.12 Spanish Mackerel (*Scomberomorus commerson*)

Spanish Mackerel (*Scomberomorus commerson*) is a highly mobile epipelagic species found in tropical and sub-tropical waters throughout the Indo-West Pacific (Newman et al., 2012; Juan-Jorda et al., 2013). Across northern Australia, Spanish Mackerel is common in inshore waters and is found near coral reefs, headlands and estuaries (Blaber et al., 1995) In the eastern Gulf of Carpentaria, Spanish Mackerel is common in highly turbid waters of shallow nearshore regions and estuaries (Blaber et al., 1995) but it generally prefers less turbid waters than Grey Mackerel. Spanish Mackerel prey on small fish (Ward and Rogers, 2003) and are also are notable predators of paneaid prawns (Salini et al., 1990; Brewer et al., 1994; Salini et al., 1998). It is fast-growing and reaches up to 70 kg, however the commercial fishery catch commonly consists of fish between 4 and 15 kg (Ward and Rogers, 2003).

Spanish Mackerel is seasonally abundant in coastal waters when large aggregations of several thousand fish can form. These aggregations are thought to be related to reproductive activity but may also be attracted by prey availability. The commercial fishery targets these aggregations given they generally occur in well-known locations such as 'Mackerel Mountain' in the southeastern GoC (Ward and Rogers, 2003).

Spawning times vary with location and repeated spawning events may occur during a season. In Western Australia, spawning occurs over August to November in the Kimberley region and over October to January in the Pilbara (Mackie et al., 2005). In Torres Strait spawning occurs over August to March and in northern Queensland (east coast) the spawning season is October to December (McPherson, 1993; Ward and Rogers, 2003). Larvae are found in inshore waters and, studies in eastern Queensland, found that highest larval abundances coincide with periods of high temperature and prey (larval fish and larvaceans (Jenkins et al., 1984)) availability (Ward and Rogers, 2003). Juveniles move into estuarine nursery habitats during the wet season (Blaber et al., 1995). Studies in eastern Queensland show juveniles move into shallow creeks, estuaries and intertidal flats, and migrate offshore in May to June (Ward and Rogers, 2003).

Data from parasites and stable isotope analyses indicate that Spanish Mackerel across northern Australia forms several stocks with limited interchange with possibly one or two subgroups in the GoC (Lester et al., 2001; Moore et al., 2003; Newman et al., 2007).

Spanish Mackerel is of great importance to commercial, charter and recreational fisheries in the GoC. The species is the main target of the Queensland L4 line fishery and is also caught in reasonable numbers in the N12 and N13 gillnet fisheries. It is less commonly caught in the N3 fishery as this species generally avoids highly turbid coastal waters.

The commercial catch in the GoC has been consistently above 150 t since 1992 and catches steadily increased from 158 t in 2000 to 320 t in 2008. Since then, the average annual catch has been 240 t (Figure 2.40). The catch from the charter fishery is only a fraction of the commercial fishery peaking at 4.8 t in 2006, but averaging around 2.5 t over the past five years. The recreational catch in the GoC in the 2010 statewide survey was estimated to be 1000 fish (Table 2.5), equivalent to around 9.5 t using an average fish weight of 9.541 kg (unpublished data, James Webley, DAFF).

Risk score justification

Spanish Mackerel differs from Grey Mackerel in that the species has less reliance on estuaries in the juvenile phase and favours less turbid waters (Figure 2.41). As such, a reduction in river flow of the Flinders and Gilbert rivers is unlikely to result in any detectable changes in population size or dynamics.

Risk scores: Consequence **0**; Likelihood **2**. **Overall risk rating**: NEGLIGIBLE

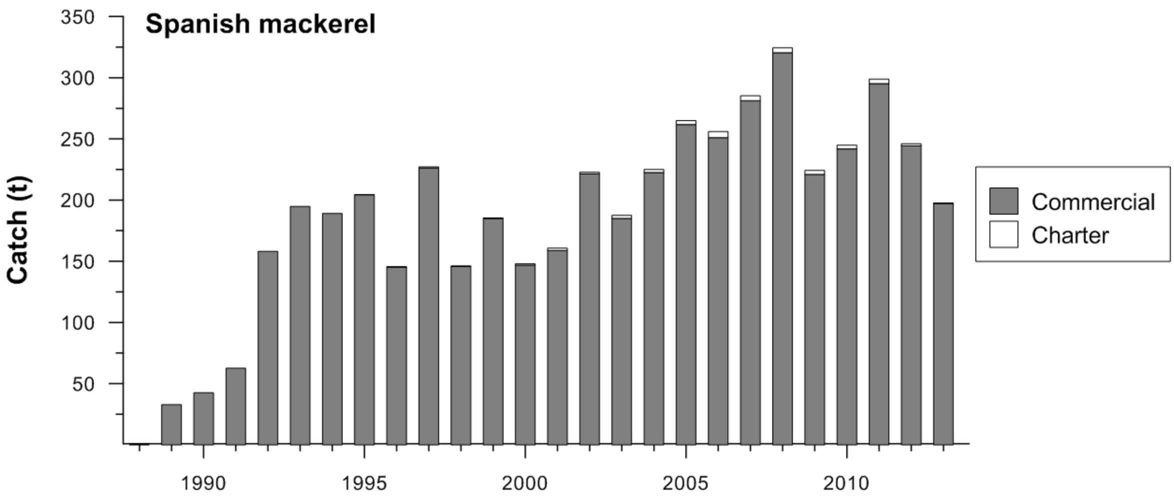

Figure 2.40. Annual catch of Spanish Mackerel (*Scomberomorus commerson*) by Queensland's commercial and charter fisheries. Note, commercial catch is combined for all net, trawl, pot and line fisheries in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

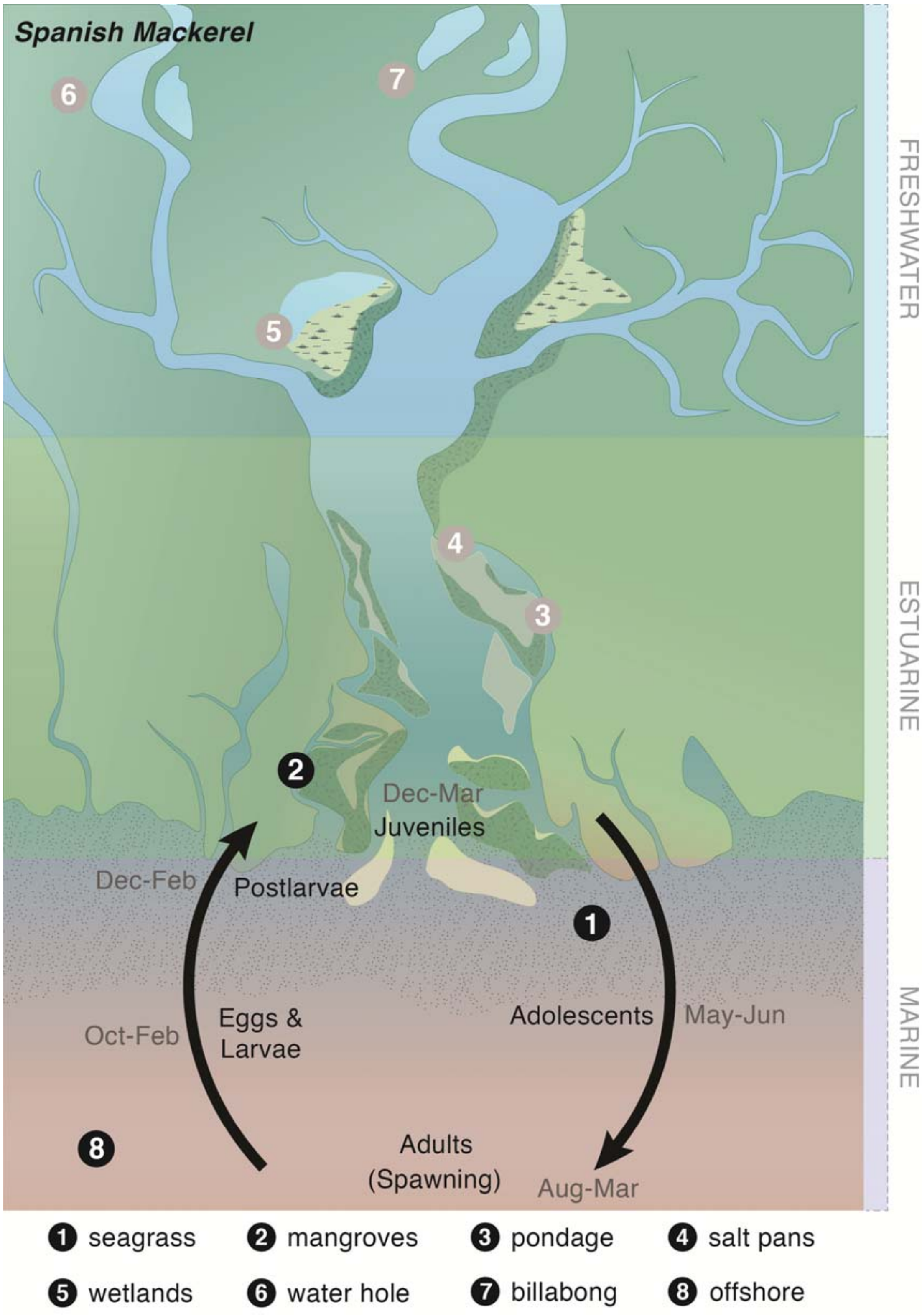

Figure 2.41. Conceptual model of the life history of Spanish Mackerel (*Scomberomorus commerson*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.13 Snappers (*Lutjanus* spp.)

'Snappers' is the generic term for species in the family Lutjanidae, which comprises over 100 species found in tropical and subtropical waters worldwide (Froese and Pauly, 2014). In Australia, this generally refers to a subset of Lutjanid species that are mostly of importance to commercial and recreational fisheries. In this review five species are included due to their similarity in life history, habitat specificity, importance to fisheries, and of highest vulnerability to altered river flows. These species include Crimson Snapper (Lutjanus *malabaricus*), Saddletail Snapper (L. *erythropterus*), Red Emperor (*L. sebae*), Golden Snapper (*Lutjanus johnii*) and goldband Snapper (*Pristopomoides multidens*).

These five Snappers are marine demersal species that are often found in association with substratum structure (e.g. reefs) in a range of depths from only a few metres to over 100m. Juveniles of these species generally use shallow coastal habitats as their primary nursery, although Red Emperor and Golden Snapper are also common in the lower reaches of estuaries, particularly around mangroves (Blaber et al., 1989; Sheaves, 1995; Tanaka et al., 2011). These species move offshore to deeper waters with increasing size, where they occupy reefs and rubble bottoms (Blaber et al., 1989; Sheaves, 1995).

Snappers are generally slow-growing and long lived across northern Australia, with several studies estimating species included in this review to live for more than 30 years (Newman et al., 2000a; Newman and Dunk, 2002; 2003; Fry and Milton, 2009). These species are gonochoristic and generally reach sexual maturity at around 40 to 50% of their maximum length (Sadovy, 1996). For example, Saddletail and Crimson Snapper in the Weipa region reach sexual maturity after 4-five years (351 mm SL) and 6 to 7 years (371 mm SL) (Fry et al., 2009), which is reasonably early in life given they have been aged to 34 and 29 years in the GoC (Fry and Milton, 2009).

Limited information is available on the reproductive biology of Snappers with the exception of Saddletail and Crimson Snapper. These species generally spawn in offshore waters around reefs but have quite different spawning periods in northern Australia, extending from July to December (Saddletail Snapper) and September to March (Crimson Snapper) (Fry et al., 2009). Therefore, their reproductive cycle is probably not exclusively linked to cues in river flows.

For most of the five species of Snappers there is a high degree of genetic population structuring, with multiple stocks existing across northern Australia and Indonesia. Otolith microchemistry and mitochondrial DNA suggests there are at least six separate stocks of Goldband Snapper across northern Australia and Indonesia (Newman et al., 2000b; Ovenden et al., 2002). Allozyme and mitochondrial DNA analysis of Saddletail Snapper suggests that the GoC supports a separate stock to the Great Barrier Reef and the northwest shelf (Elliott, 1996), while Crimson Snapper appears to also have a similar genetic population substructure (Blaber et al., 2005).

Movements of Snappers are not well understood, except the general pattern than most species appear to move considerable distance offshore to spawn and/or to complete the adult phase of their life history (e.g. Russell and McDougall, 2005). Although many fish representing each Snapper species have been tagged, generally in recreational fishing tagging programs, few recaptures have been recorded, possibly due to the high mortality of released fish as a result of barotrauma (Sumpton et al., 2010).

Snappers are one of the most commercially important species in the GoC with Crimson Snapper and Saddletail Snapper having the 7th and 9th highest annual catches in the past five years, which has almost exclusively come from the fish trawl fishery. The annual commercial catch of Snappers in the GoC rapidly increased from 2002 to a peak in 2008 and 2009 of around 840 t, where the vast majority of the catch was comprised of Crimson Snapper (46%) and Saddletail Snapper (30%). Recently, Snapper catches have significantly declined from 630 t in 2011 to just 7 t in 2013, however, this is a likely result of a dramatic decrease in fishing effort (Figure 2.42).

Snapper are also an important recreational species across northern Australia, being taken in the charter and recreational fishery. The annual charter catch for the five species combined was less than 10 t over the past five years, with Red Emperor comprising around 50% of this catch. No recreational catch estimate is available for Red Snappers in 2010, although 1000 Golden Snapper were estimated to be caught (Table 2.5).

Risk score justification

The five species of Snapper assessed complete the majority of their life cycle offshore (including spawning) and have very little reliance upon estuaries to complete their life cycle (Figure 2.43). An exception may be the use of mangrove areas by Red Emperor and Golden Snapper, but these species also use other shallow coastal habitats. As such, a reduction in river flow of the Flinders and Gilbert rivers is unlikely to result in any detectable changes in population size or dynamics.

Crimson, Saddletail and Goldband Snappers

Risk scores: Consequence **0**; Likelihood **3**. **Overall risk rating**: NEGLIGIBLE

Red Emperor and Golden Snapper

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

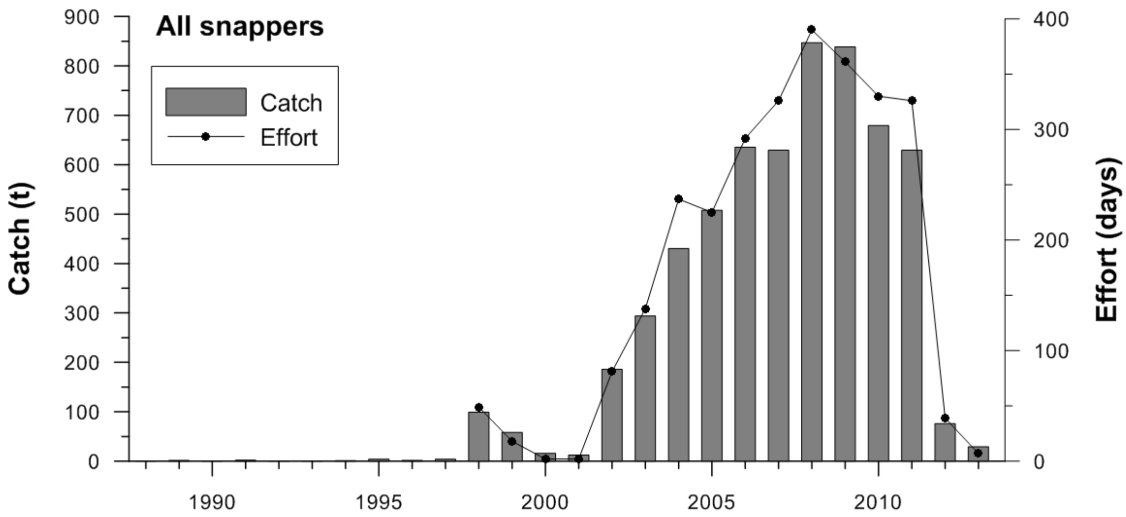

Figure 2.42. Annual combined catch of "Snappers" Crimson Snapper (*Lutjanus malabaricus*), Saddletail Snapper (*L. erythropterus*), Red Emperor (*L. sebae*), Golden Snapper (*L. johnii*) and Goldband Snapper (*Pristopomoides multidens*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Fishing effort (in days) by the developmental finfish trawl fishery is also shown. All data supplied by Fisheries Queensland.

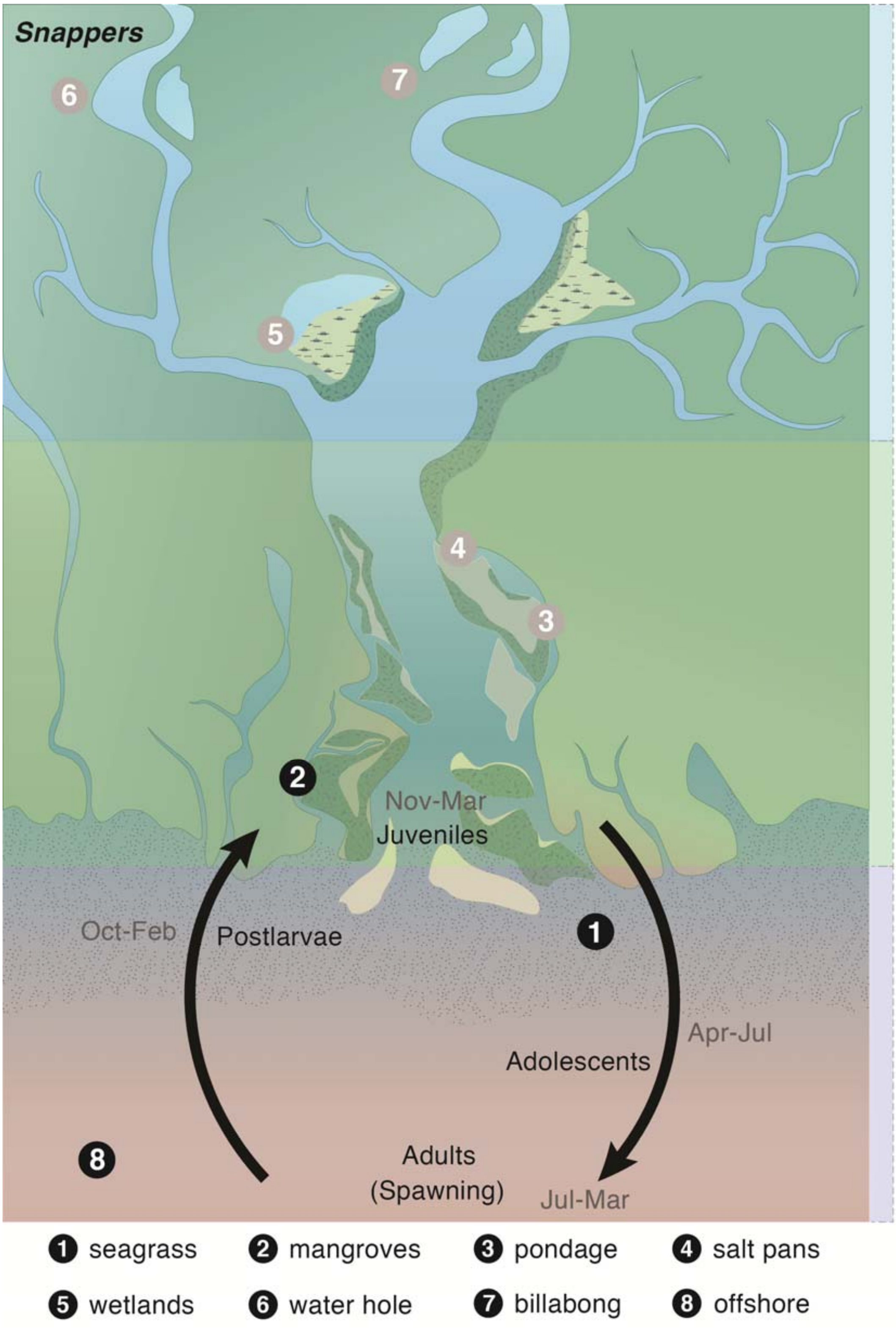

Figure 2.43. Conceptual model of the life history of "Snappers" – namely Crimson Snapper (*Lutjanus malabaricus*), Saddletail Snapper (*L. erythropterus*), Red Emperor (*L. sebae*), Golden Snapper (*L. johnii*) and Goldband Snapper (*Pristopomoides multidens*) – illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.14 King Threadfin (*Polydactylus macrochir*)

King Threadfin (*Polydactylus macrochir* previously *P. sheridani*) is endemic to northern Australia, southern Papua New Guinea and Irian Jaya (Motomura et al., 2000). King Threadfin is an important species for fisheries in the Gulf (Halliday and Robins, 2005). King Threadfin has a potential lifespan of at least 22 years, and is a protandrous hermaphrodite. They reach maturity between two and five years and most individuals change sex from male to female at between six and ten years.

King Threadfin occurs in tropical and sub-tropical turbid coastal waters, estuaries and mangrove creeks (Blaber et al., 1995). Juvenile King Threadfin likely enter estuaries during the wet season when prawns and other prey species are seasonally abundant (Halliday et al., 2008) King Threadfin is carnivorous, feeding on primarily small fish but are also notable predators of penaeid prawns (Salini et al., 1990; Blaber et al., 1995).

Adults probably spawn in inshore coastal waters and lower parts of estuaries (Halliday and Robins, 2005; Welch et al., 2014). High salinity (>32‰) is important for survival of pelagic eggs and spawning occurs away from the mouths of rivers where salinity levels may be lower (Robins and Ye, 2007; Halliday et al., 2008; Welch et al., 2014). The length of the spawning season and timing of peak spawning varies among regions. Spawning occurs over October to January (spring and summer) on the east coast of Queensland, between late winter and spring in the GoC and from October to March in the NT (Halliday et al., 2008). Juveniles move into shallow nearshore waters and estuaries. On the east coast of Queensland, early juveniles are reported in estuaries between December and May and across a wide range of salinities but not in freshwater (Halliday et al., 2008; Welch et al., 2014). King Threadfin juveniles are thought to restrict their use of estuarine habitats to permanent water areas in the main channels and tributaries of creeks and rivers (Halliday et al., 2008). Wetland habitat extent and connectivity is likely to be important in supporting King Threadfin production (Meynecke et al., 2008)

It is proposed that high flow enhances juvenile survival through enhanced productivity of coastal waters, increased the extent of lower-salinity habitat and/or reduced predation risk in higher turbidity waters (Halliday et al., 2008). However, no relationships were found among rainfall or flow with commercial catch or year-class strength from the Mitchell, Flinders, Roper and Daly estuaries; movement along coastlines may confound potential relationships (Halliday et al., 2012). In the Fitzroy River on the east coast of Queensland, the year-class strength of King Threadfin commercial catches are positively correlated with flow and coastal rainfall over spring and summer (Halliday et al., 2008). The large decadal variability in flow this sub-tropical estuary, compared to the more predictable monsoonal rainfall in the tropics, could account for these positive correlations (Halliday et al., 2012).

It is suggested that salinity changes may stimulate the movement of King Threadfins out of estuaries thus increasing vulnerability to the fishery (Halliday and Robins, 2005; Robins and Ye, 2007). In winter, adults ocur in the middle to upper reaches of estuaries, when saline waters intrude up the estuary but leave the estuaries with the onset of wet season high flows (Halliday et al., 2008).

Considerable variation in growth rates, age at sex change and maximum age for fish sampled across northern Australia suggesting that a number of geographically and/or reproductively distinct groups exist (Moore et al., 2012b). However, no significant differences were found between fish sampled from south-east GoC (Flinders River and Spring Creek within the Gilbert River catchment). This, coupled with evidence from genetic, otolith, stable isotope and parasite analyses, indicates that King Threadfin in south-east GoC waters form one discrete stock of post-recruit fish with little adult or larval interchange from other populations (Newman et al., 2010b; Welch et al., 2010; Horne et al., 2012; Moore et al., 2012b; Moore et al., 2012a). Further, a large proportion of small, young females and earlier age at sex change in fish sampled from eastern Gulf waters, coupled with temporal instability in genetic structure, suggest King Threadfin may be heavily fished in this area (Welch et al., 2010; Horne et al., 2012; Moore et al., 2012b).

King Threadfin is a highly important species to commercial, charter and recreational fisheries in the GoC. Although catch data from Queensland fisheries were not available by fishery, most of the commercial catch comes from the commercial N3 gillnet fishery. The commercial catch of King Threadfin in the GoC has been very stable since 1989 with peaks in 1991 and 2001 of 486 t and 473 t, respectively (Figure 2.44). Catches have been stable at around 300 t since 2003, with only 2013 recording a catch lower than 200 t. The recreational catch in 2010 was estimated to be 7000 fish (Table 2.5), equivalent to around 21 t, assuming an average fish weight of 3 kg. Interestingly, the catch of King Threadfin by the charter fishery has been very low in the GoC, averaging less than 1 t over the past five years.

Risk score justification

Abundance of King Threadfin may be negatively impacted by reduced flows as a result of reduced productivity of coastal and estuarine waters and seasonal prey availability (e.g. panaeid prawns that are highly dependent on flows), and the absence of high flows to stimulate the movement of adolescent King Threadfin out of the estuaries. Growth and survival of juvenile King Threadfin may be negatively impacted by a reduction in wet season flow through loss of access to water channels (extent and connectivity) in estuaries and wetlands (Figure 2.45). The high population subdivision in the southeast GoC increase the likelihood that any impacts from reduced flows will impact an entire breeding stock.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

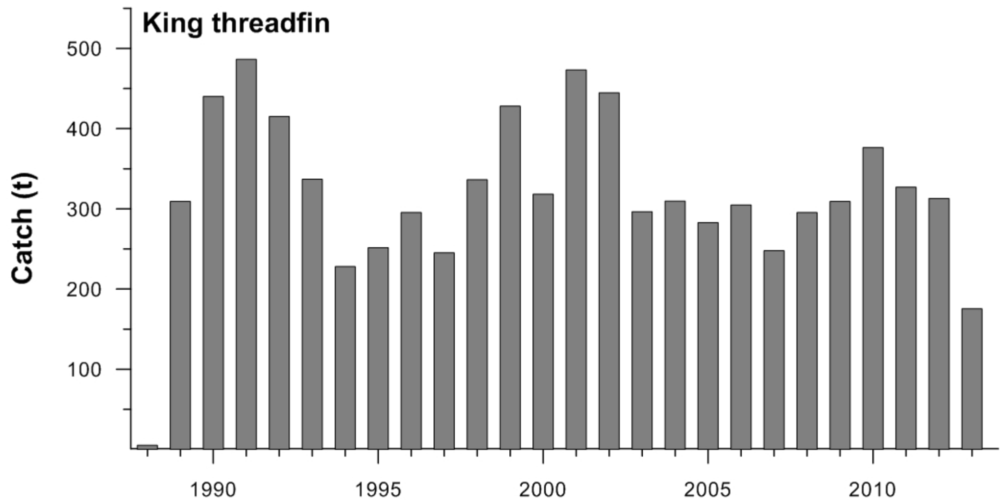

Figure 2.44. Annual catch of King Threadfin (*Polydactylus macrochir*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

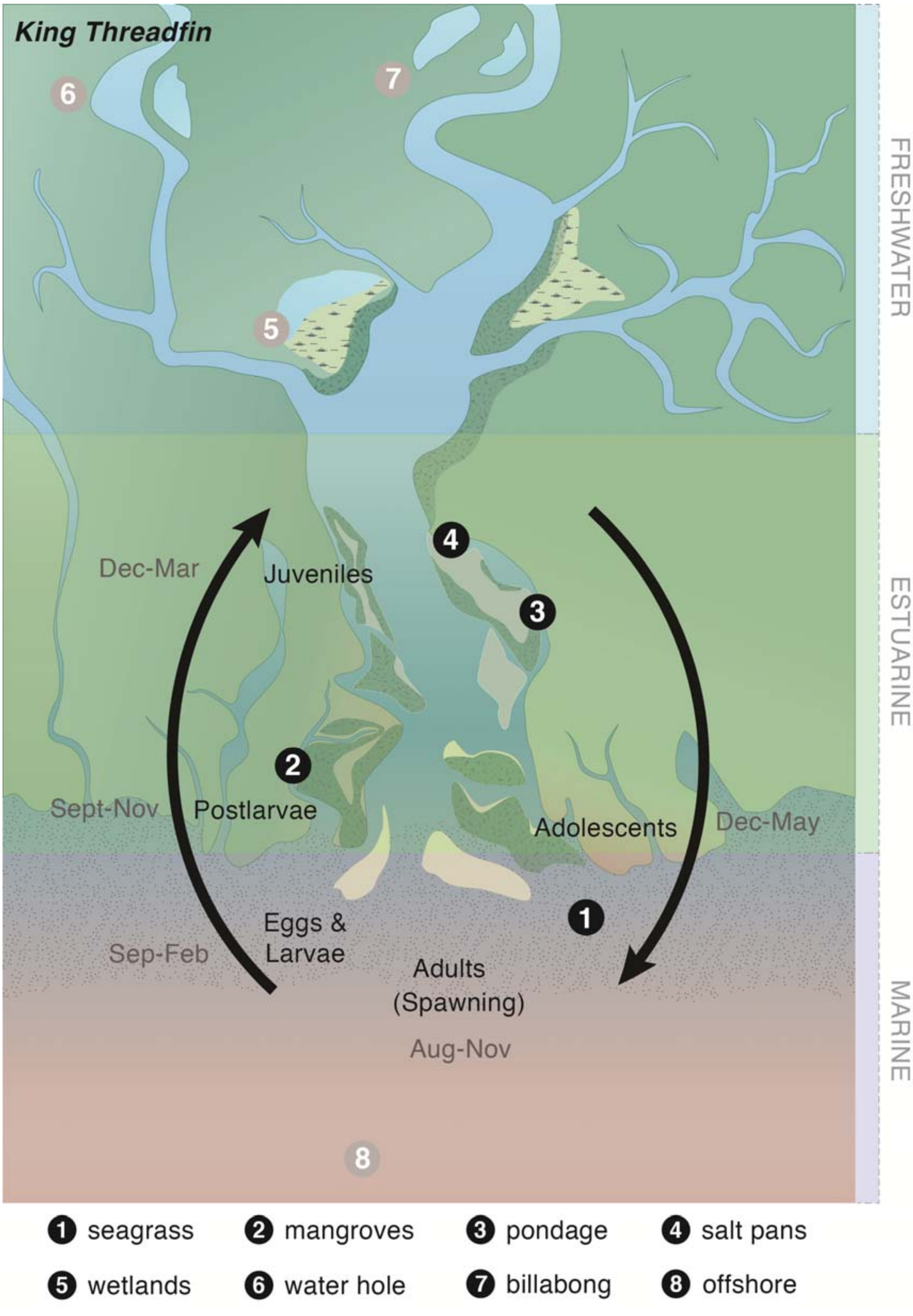

Figure 2.45. Conceptual model of the life history of King Threadfin (*Polydactylus macrochir*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.15 Blue Threadfin (*Eleutheronema tetradactylum*)

Blue threadfin (*Eleutheronema tetradactylum*) is distributed throughout the tropical and subtropical regions of the Indo-West Pacific and is common across northern Australia. Although the species can tolerate an extremely wide range of salinities from 0 to 39.0 PSU in the GoC (Cyrus and Blaber, 1992), it generally prefers shallow turbid marine and estuarine waters where salinity is more stable. It is an important predator in these habitats feeding mainly on crustaceans and fish, and in particular, it is among the largest consumer of commercially important penaeids in the GoC (Haywood et al., 1998; Salini et al., 1998).

Blue Threadfin in northern Australia is reasonably fast growing, lives for at least seven years (Bibby and McPherson, 1997). As with several GoC estuarine-dependent species, the Blue Threadfin is a protandrous hermaphrodite maturing first as males and reach sexual maturity in around two to three years (Ballagh et al., 2012), and later changing to females in the GoC (McPherson, 1997). However, there is significant variability in growth and reproductive dynamics of Blue Threadfin between estuaries in northern Australia. Fish from the southeast GoC rivers (Archer and Love) tended to attain a smaller length-at-age, exhibited the slowest growth rates, but matured at an early age (1 to 2 years) (Ballagh et al., 2012). In the south-east GoC, Blue Threadfin spawn between July and October (McPherson, 1997) in coastal waters away from the direct influence of fresh water river discharge (Garrett and Williams, 2002). Juveniles recruit to inshore tidal flats and the lower reaches of estuaries areas (Russell & Garrett 1983).

Movements of Blue Threadfin are not well understood. Many fish have been tagged by recreational fishers, but most recaptured fish have moved short distances, although some large-scale movements of up to 150 km have been documented (Sawynok, 1991). Four recent studies have shown there to be a high level of population subdivision in Blue Threadfin across northern Australia using evidence from genetics (Horne et al., 2011), parasites (Moore et al., 2011), life history characteristics (Ballagh et al., 2012) and otolith microchemistry (Newman et al., 2011). These studies suggest that individual estuaries separated by as little as 15 km may support individual populations. Similar population subdivision is also evident on the Queensland east coast estuaries as determined by parasites and tagging (Zischke et al., 2009).

Blue Threadfin is a highly important species to commercial, recreational and Indigenous fisheries in the GoC. Although catch data from Queensland fisheries are not available by fishery, most of the commercial catch comes from the commercial N3 gillnet fishery. It is also a very important recreational species being taken in the charter and recreational fishery.

The commercial catch of Blue Threadfin in the GoC has been variable since 1988, averaging around 80 t. Catches peaked in 2004 (126 t), but have generally declined to around 40 t in 2012 and 2013 (Figure 2.46). The charter catch has remained stable since 1998 at around 7 t but has increased since 2005 to around 10 t per year. The recreational catch in 2010 was estimated to be 17,000 fish (Table 2.5), equivalent to around 34 t (assuming an average weight of 2 kg), or around half of the commercial catch.

Risk score justification
The species has a high dependence upon estuaries to complete its life cycle (Figure 2.47) and river flows are most likely a cue for spawning migration. Although the species appears to be relatively fast growing and early to mature, its protandrous hermaphroditism, the high variability in life history parameters, and genetic subdivision between estuaries separated by only tens of kilometres is a significant cause for concern for the viability of this species under altered flow regimes.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

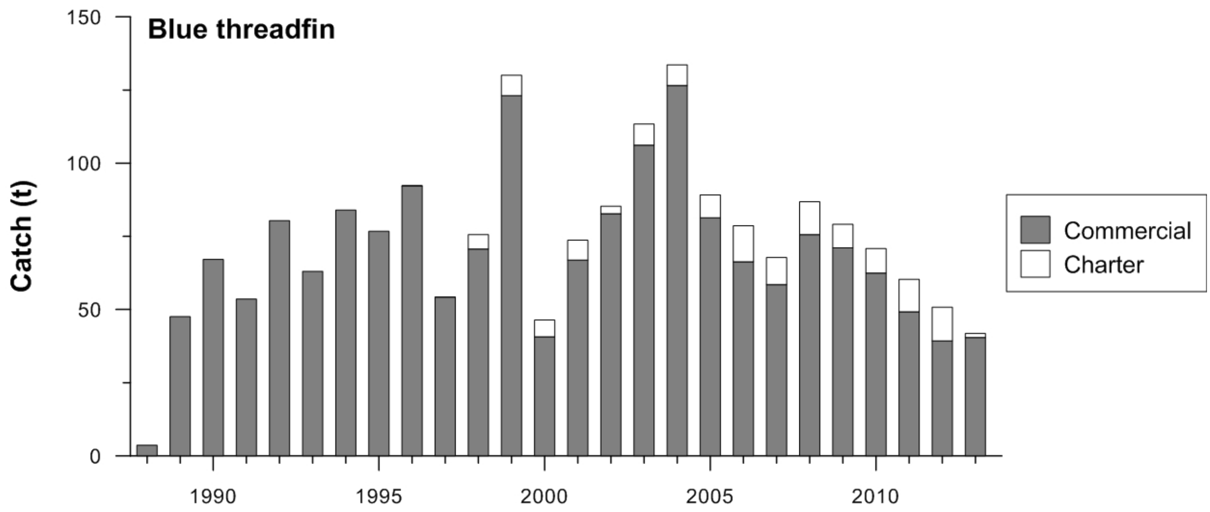

Figure 2.46. Annual catch of Blue Threadfin (*Eleutheronema tetradactylum*) by Queensland's commercial and charter fisheries in the Gulf of Carpentaria. Note, commercial catch is combined for all net, trawl, pot and line fisheries. Data supplied by Fisheries Queensland.

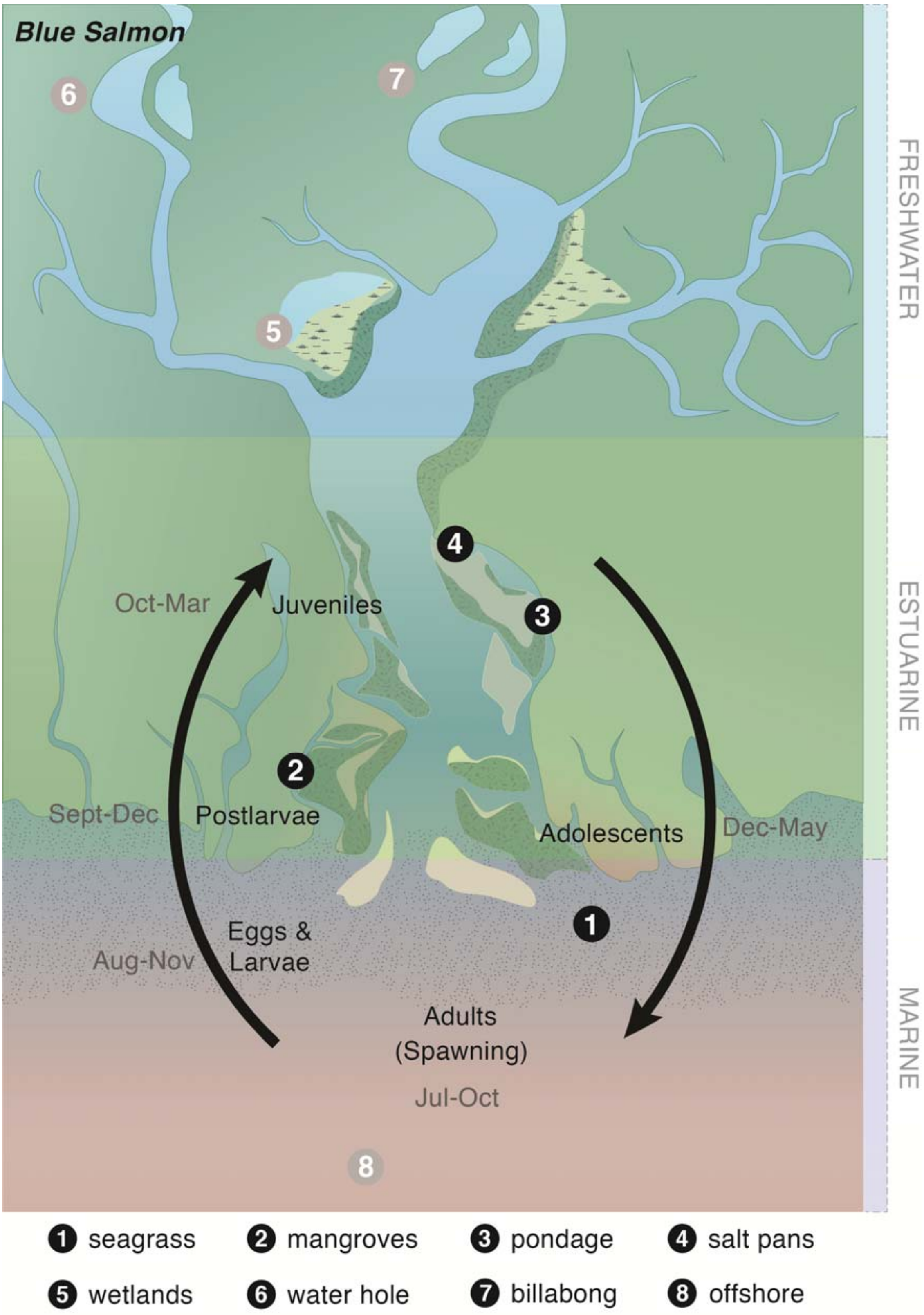

Figure 2.47. Conceptual model of the life history of Blue Threadfin (*Eleutheronema tetradactylum*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.16 Grunter (*Pomadasys kaakan*)

Grunter (*Pomadasys kaakan*), or Barred Javelin, is a coastal demersal species distributed widely throughout the Indo-West Pacific, from the east coast of Africa to southern Queensland, Australia (Froese and Pauly, 2014). It is widely distributed throughout the GoC but is more abundant in inshore shallow waters and estuaries (Blaber et al., 1994a; Blaber et al., 1995) where it can tolerate a range of salinities (0 to 39.0 PSU) and turbidities (1.2 to 21.0 NTU) (Cyrus and Blaber, 1992).

The life history of Grunter has been little studied, with only one study in Australian waters (Szczecinski, 2012). Most biological knowledge comes from the waters of Iran (Fakhri et al., 2011; Falahatimarvast et al., 2012), Pakistan (Majid and Imad, 1991) and Kuwait (Al-Husaini et al., 2001; Al-Husaini et al., 2002).

Grunter grow to a maximum length of 80 cm TL (Froese and Pauly, 2014). In northeastern Australia, the Grunter is slow growing, reaching 50% of its maximum length by about age four years, and lives to at least 15 years (Szczecinski, 2012). However, in the waters of Kuwait, the validated maximum age for Grunter was 36 years (Al-Husaini et al., 2002) The sex ratio of fish is significantly skewed towards females, although there is no evidence of ontogenetic sex inversion from macroscopic staging of gonads. Grunter mature early in life, with 50% of females mature by the age of two to three years (280 to 319 mm TL) (Szczecinski, 2012). This is significantly smaller than the length at 50% maturity estimate of 477 mm TL – based on histology – for female fish in the waters of Iran (Falahatimarvast et al., 2012).

Grunter spawn in northeastern Australia between August and November, although a ~20% of females are in spawning condition all year (Szczecinski, 2012). In the GoC, the exact timing of spawning is uncertain, but it may occur between September and March (Robins and Ye, 2007; Welch et al., 2014). Spawning is thought to occur outside of estuaries in marine waters as eggs require high salinity for survival (Robins and Ye, 2007). Juveniles Grunter recruit to a range of shallow nearshore habitats including estuaries, seagrass beds and prawn trawl grounds (Blaber et al., 1989; Blaber et al., 1990; Blaber et al., 1992; Blaber et al., 1994a; Blaber et al., 1995).

Little is known of the movements or genetic population structure of Grunter in Australian waters. However, given its widespread distribution, occupation of a range of habitats and high salinity and turbidity tolerance, it is likely that Grunter exists as a single stock throughout the GoC.

Grunter is a generalist feeder, consuming a variety of fish and benthic invertebrates including penaeid prawns in the Norman River (Salini et al., 1998). It has been suggested that aggregations of Grunter around river mouths in winter months are in response to seasonal concentrations of prey, possibly White Banana Prawns.

Grunter is a highly important species to commercial, recreational and Indigenous fisheries in the GoC. Although catch data from Queensland fisheries are not available by fishery, most of the commercial catch comes from the commercial N3 gillnet fishery. The commercial catch of Grunter in the GoC has been consistently above 10 t since the early 1990s with the catch peaking at 52 t in 2004 (Figure 2.48). In the past five years, the annual catch has averaged 14 t. The annual charter catch peaked at 3.6 t in 1997, but has only averaged 350 kg per year over the past five years.

Grunter was the third most important species (by number) to recreational fishers in the study region in the 2010 statewide survey. Recreational fishers caught an estimated 38,000 fish (Table 2.5), of which an estimated 54% were released (QFISH online database). The total catch, assuming an average weight of 1.5kg, equates to around 57 t. This is one of the few species where the recreational catch exceeds the commercial catch. Because Grunter inhabits inshore waters and estuaries, it is also likely to be caught by Indigenous fishers (Zeller and Snape, 2006).

Risk score justification

Grunter uses estuaries as a nursery ground for juveniles and as an important feeding ground for adults (Figure 2.49). However, its extensive use of marine waters for feeding and spawning may suggest that a reduction in river flows may not have a large impact on the population size or structure of this species. However, given the species' slow growth, longevity and late maturation, even small population declines caused by reduced river flows may take considerable time to rebuild.

Risk scores: Consequence **2**; Likelihood **3**. **Overall risk rating**: MODERATE

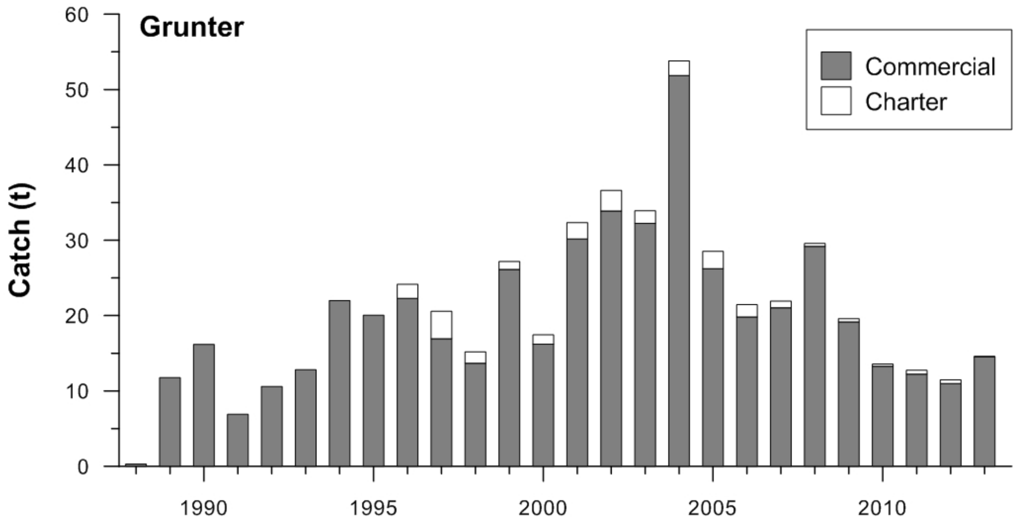

Figure 2.48. Annual catch of Grunter (*Pomadasys kaaken*) by Queensland's commercial and charter fisheries in the Gulf of Carpentaria. Note, commercial catch is combined for all net, trawl, pot and line fisheries. Data supplied by Fisheries Queensland.

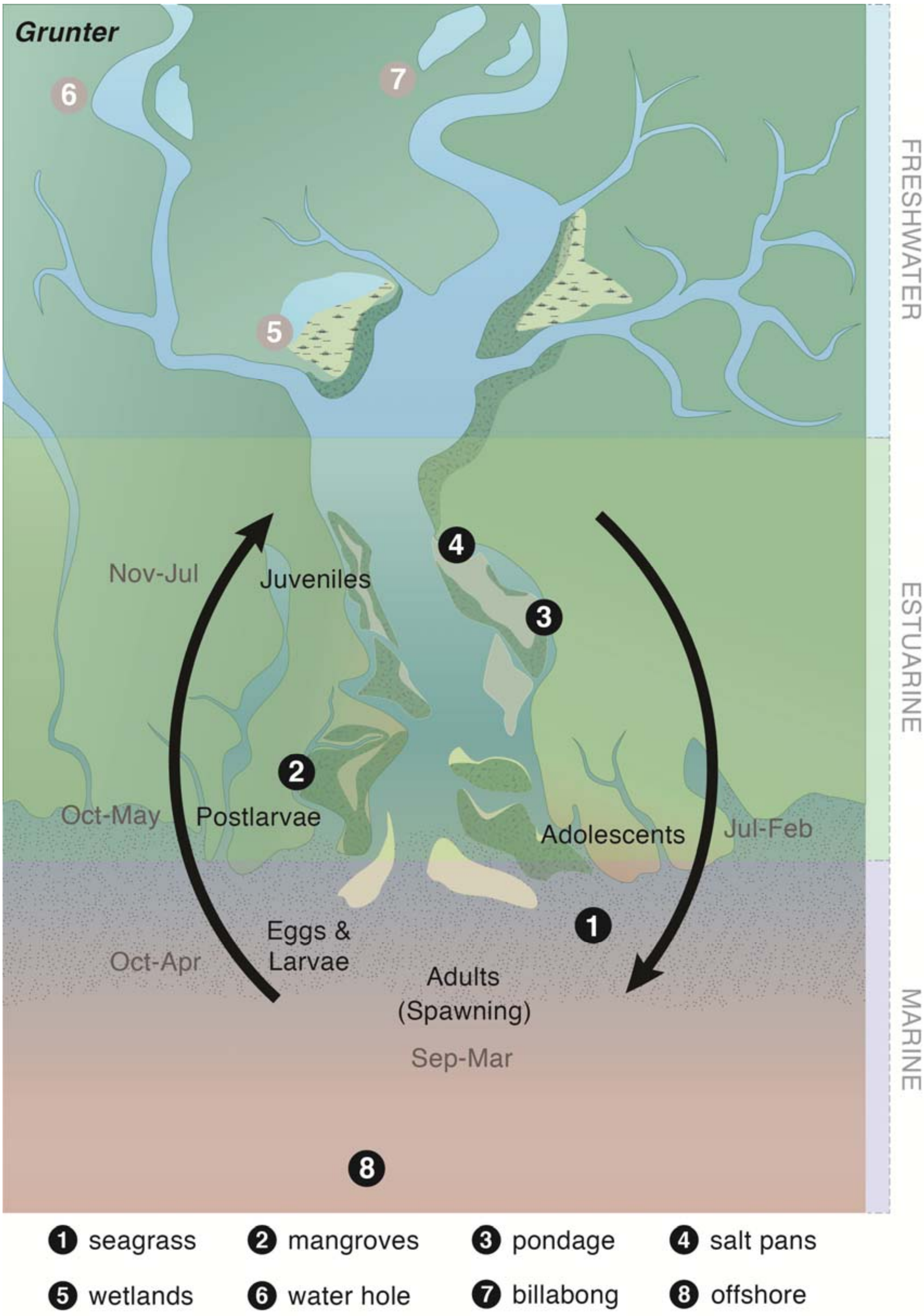

Figure 2.49. Conceptual model of the life history of Grunter (*Pomadasys kaaken*) illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.17 Talang Queenfish (*Scomberoides commersonianus*)

The Talang (or Giant) Queenfish (*Scomberoides commersonianus*) (family Carangidae) is the largest of four pelagic species in the genus found throughout the Indo-West Pacific and is distributed in Australia from northern New South Wales to Exmouth, Western Australia. The species can be found in clear to highly turbid waters, but generally prefers higher salinities in the lower reaches of estuaries around reefs in marine waters.

An ageing study using otoliths showed that Talang Queenfish in the eastern GoC is slow-growing, reaching around 50 cm TL after three years and living for at least 11 years (Griffiths et al., 2006). Little is known of the population structure or movements of Talang Queenfish. Available tagging data show fish can move up to 200 km, but most recaptures of tagged fish occur within 50km of the release point (Suntag unpublished data). Therefore, Talang Queenfish probably comprise a single stock throughout the GoC.

The Talang Queenfish is gonochoristic with a 1:1 male to female sex ratio. They are relatively late maturing with 50% of females reaching sexual maturity at age four to five years in the GoC (Griffiths et al., 2006). The species has an extended spawning season in the GoC from September to March. Little is known of the spawning locations of Talang Queenfish, however, the capture of large ripe female fish approximately 7 to 10 nm from Duyfken Point off Weipa suggests it spawns offshore.

Juveniles Queenfish recruit to estuaries and shallow nearshore habitats following the wet season, where they can be found in large schools. Blaber et al. (1989) estimated that the standing biomass of Queenfish in Embley estuary was 2.3-4.1 gm^{-2} and represented 25-32% of the total fish biomass. Ecologically, the species is one of the most important predators in coastal and estuarine habitats throughout the GoC. Its feeding ecology has been well studied in the GoC including Weipa (Salini et al., 1990; Haywood et al., 1998), Groote Eylandt (Brewer et al., 1995) and the Norman River (Salini et al., 1998), where they consume a range of benthic and pelagic prey. In particular, this species is one of the highest consumers of penaeids, including several commercially important species, such as White Banana Prawns, before they move from the estuaries into the GoC (Haywood et al., 1998).

Talang Queenfish is of importance to commercial, charter and recreational fisheries in the GoC. Although it is generally a byproduct species in the N3 and N12/13 gillnet fisheries, it is an important recreational sportfish in the region. The commercial catch in the GoC has been variable since 1988, but averaged around 20 t per year. Catches peaked in 2004 (35 t) and 2008 (32 t), but catches have declined since 2010 to 7 t in 2013 (Figure 2.50). The charter catch has remained stable at around 1 to 2 t since 1996 (Figure 2.50), while the recreational catch in 2010 was estimated to be 4000 fish (Table 2.5), equivalent to around 8 t if assuming an average weight of 2 kg.

Risk score justification

The species has a high dependence on estuaries, both as a nursery ground for juveniles and as an important feeding ground for adults (Figure 2.51). However, its extensive use of marine waters for feeding and spawning suggests that a reduction in river flows may not have a large impact on the population size or structure of this species. However, given the species' slow growth and late maturation, even small population declines may take considerable time to rebuild.

Risk scores: Consequence **2**; Likelihood **4**. **Overall risk rating**: MODERATE

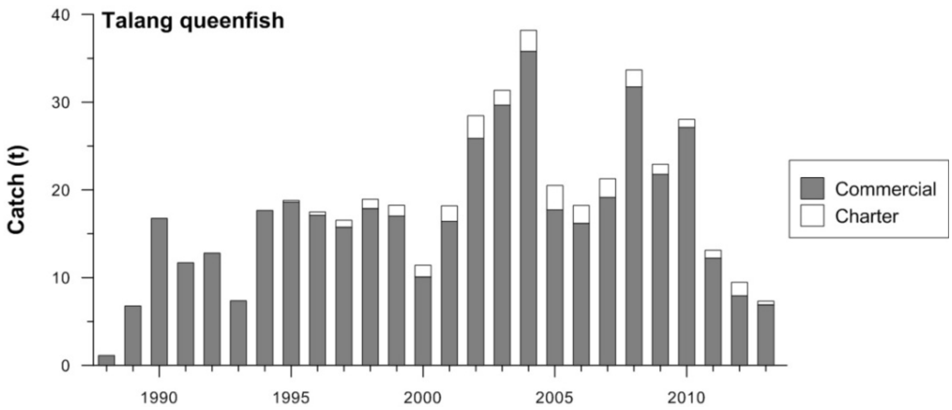

Figure 2.50. Annual catch of Talang Queenfish (*Scomberoides commersonianus*) by Queensland's commercial and charter fisheries in the Gulf of Carpentaria. Note, commercial catch is combined for all net, trawl, pot and line fisheries. Data supplied by Fisheries Queensland.

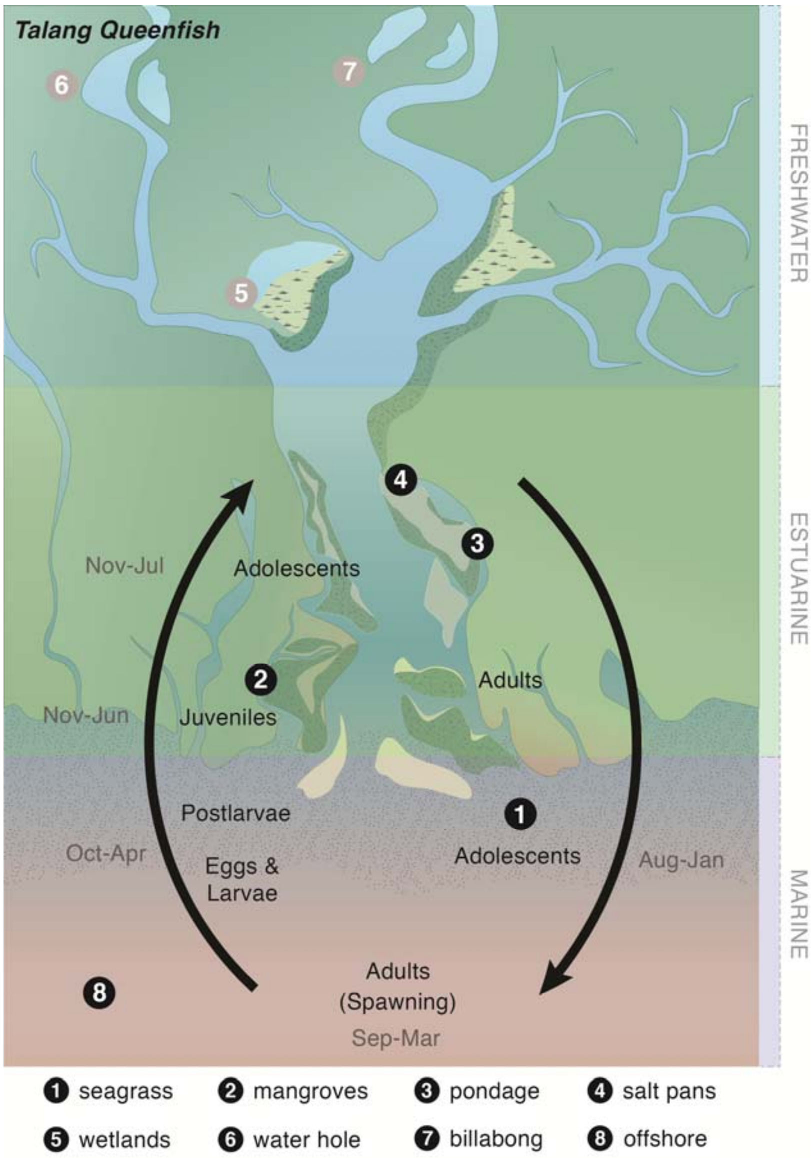

Figure 2.51. Conceptual model of the life history of Talang Queenfish (*Scomberoides commersonianus*) illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.18 Pikey Bream (*Acanthopagrus berda*)

Pikey Bream (*Acanthopagrus berda*) is an estuarine-dependent diadromous demersal species that inhabits coastal, estuarine and freshwaters across northern Australia (Blaber et al., 1989). Its distribution extends throughout the Indo-Pacific region and it is particularly abundant in South African estuaries where most of the biological and ecological knowledge for the species has been derived.

Pikey Bream is a small, slow-growing species reaching at least 16 years of age in both northern Australia (Tobin, 1998) and South Africa (James et al., 2003). There is considerable variation in growth rates of fish between estuaries in northern Australia coast as determined by otoliths (Tobin, 1998) and tagging (Suntag, unpublished tagging data). This suggests there may be subdivision of the population to the extent that individual populations may exist in each estuary.

Pikey Bream has a similar life history to Barramundi, in particular undergoing protandrous sex inversion whereby small fish are generally males, transitioning to female with age. Sexual maturity occurs late in life (3 to 4 years) (James et al., 2008). Spawning takes during the dry season between June and September along the east Queensland coast (Tobin, 1998) – presumably when river flows are lowest – where the sex ratio is highly biased towards males. Although the exact spawning location is unknown in Australia, in the Kosi Estuary in South Africa, Pikey Bream spawns at the mouth of the estuary and move back up river into estuarine and freshwater (Garratt, 2012). It was noted that degradation of estuarine habitats in the Kosi via reduction of river flows is likely to significantly disrupt the spawning migration and subsequent population size of the species in the area.

In GoC rivers, juvenile and adult Pikey Bream co-occur in a wide range of habitats from shallow marine areas, estuaries, to freshwater including water holes and billabongs (Blaber et al., 1989; Salini et al., 1998). This is a result of the species' high tolerance to a wide range of salinities from 0 to 35.3 PSU (Cyrus and Blaber, 1992). Its abundance and nearshore distribution make Pikey Bream an important target of recreational fisheries. In the 2010 statewide recreational fishing survey, Pikey Bream was the second most important species caught (49,000 fish) in the southwestern GoC (Table 2.5), of which 49% of fish were reported to have been released (QFISH online database). In contrast, Pikey Bream is not considered to be of importance to any commercial fisheries in the GoC with the average annual catch being 273kg in the five year period between 2009 and 2013. Similarly, the species is of low importance to charter fishery with the average annual catch in the GoC being 310 kg in the past five years (DAFF unpublished logbook data).

Risk score justification

The species is estuarine dependent to complete its life cycle and uses marine to freshwater (Figure 2.52). Its slow growth, late maturity and protandrous sex change suggest that even small population declines are likely to take considerable time to rebuild. Furthermore, the highly variable growth rates documented between rivers using both otoliths and tagging suggest the population may be highly subdivided whereby the Flinders and Gilbert rivers may support individual populations.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

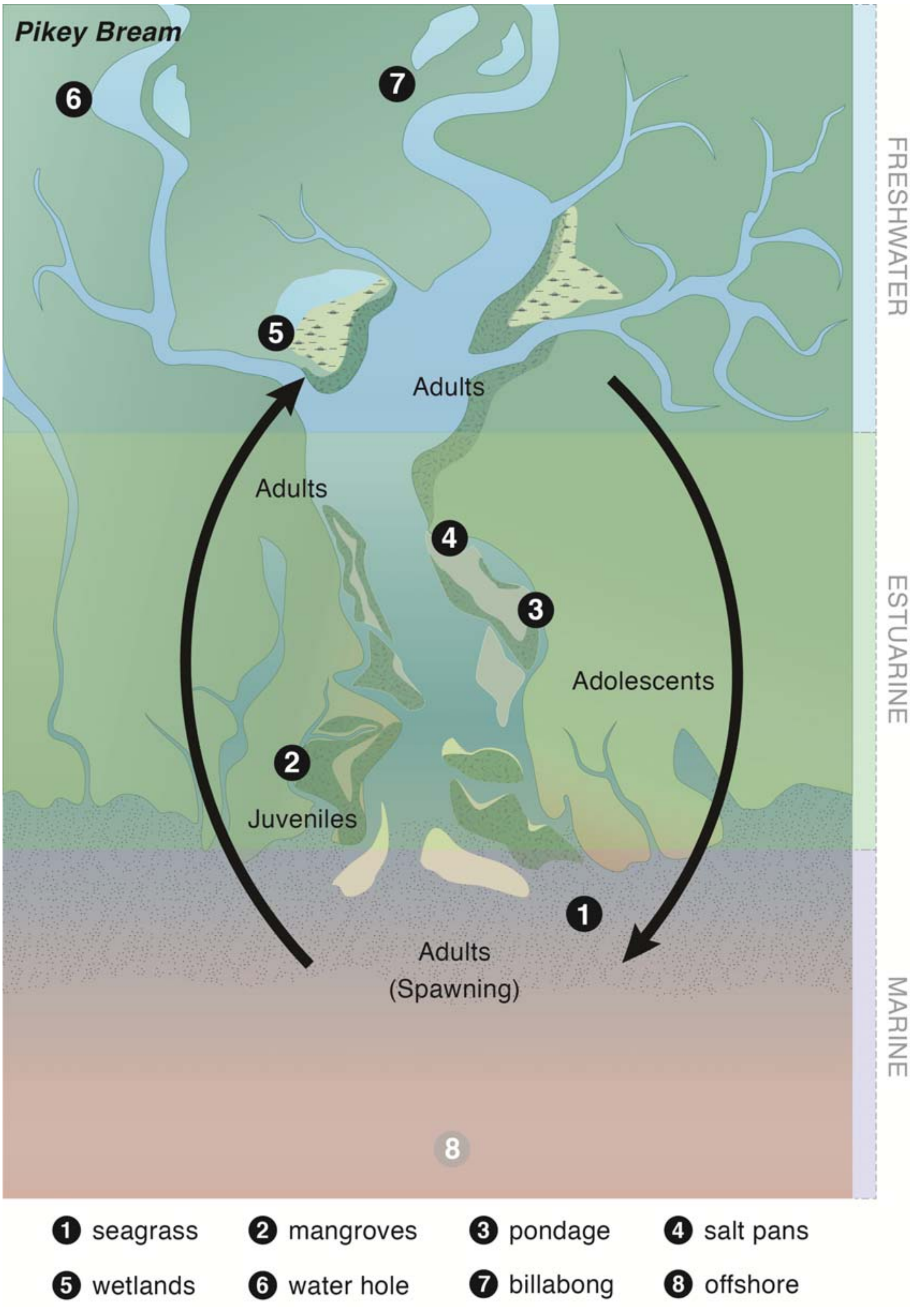

Figure 2.52. Conceptual model of the life history of Pikey Bream (*Ancanthopagrus berda*) illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.19 Black Jewfish (*Protonibea diacanthus*)

Black jewfish (*Protonibea diacanthus*) is a large fish widely distributed in coastal, tropical waters that is likely to migrate seasonally (Phelan et al., 2008). It is a long lived fish (at least 12 years) but grows rapidly (Phelan et al., 2008). Adults are found on near-shore reefs, whereas juveniles occur in coastal embayments and estuaries (Welch et al., 2014). In northern Australia, black Jewfish form large aggregations, up to several thousand fish, at well defined locations in turbid inshore waters that are fished by commercial, traditional and recreational fishers; heavy exploitation of aggregations can be detrimental for local stocks (Bibby et al., 1992; Phelan et al., 2008). Aggregations occur from August to January in NT (Phelan, 2008; Semmens et al., 2010) and April to September off Cape York (Phelan et al., 2008). Evidence suggests little movement of fish between aggregations and sites, suggesting high site fidelity and thus separate adult populations (Phelan, 2008; Semmens et al., 2010).

It is likely these aggregations are related to spawning and/or feeding; development of gonads is coincident with aggregation season with anecdotal evidence of ripe eggs in fish caught from aggregations (Phelan, 2008; Phelan et al., 2008). Reproductive season in the NT extends from August to January with a peak in December (Phelan, 2008; Semmens et al., 2010). Spawning in both the NT and Indian waters occurs during the summer monsoon season, when coastal waters are warm, and also turbid due to high river flow (Phelan, 2008; Semmens et al., 2010).

Annual aggregations may also be related to prey availability, but there is no evidence to date to support this from Australian waters (Phelan et al., 2008). However, adults tend to occupy near shore reefs, and juveniles coastal embayments and estuaries, therefore flow is likely to influence food availability hence growth and survival. In the NT, its feeding behaviour appears to have some relation to the state of the tides and hence water flow (Semmens et al., 2010). Black Jewfish feed on fish, gastropods, prawns and other crustaceans (Phelan et al., 2008). Linkages among recruitment, abundance and flow are poorly understood but likely to be important given the life cycle of black Jewfish (Welch et al., 2014).

Black Jewfish is a common byproduct species in the N3 inshore gillnet fishery and often targeted by charter and recreational fishers when targeting various species of reef fish, such as Snappers. The commercial catch in the GoC has been very stable since 1989, with peaks in 1991 and 2001 of 486 t and 473 t, respectively (Figure 2.53). Catches have been stable at around 2 to 3 t from 1997 to 2006, after which catches peaked at 9 t in 2009. The average catch in the past five years was 6 t. Jewfish is an extremely important species in the recreational fishery, representing the ninth most caught species in the 2010 statewide survey, where an estimated 12,000 fish were caught (Table 2.5). Using an average fish weight of 5kg, this equates to around 60 t, more than six times the commercial catch. Interestingly, the catch of Black Jewfish by the charter fishery is very low in the GoC, averaging only 150 kg per year over the past five years.

Risk score justification

A reduction in wet season flow is likely to reduce extent and connectivity of channel habitat as well as prey availability for Black Jewfish, and therefore affect growth and survival of both adults and juveniles given their occurrence in nearshore and estuarine waters (Figure 2.54). Reduced flows at this time may also reduce the efficacy of cues to form large aggregations that this species forms at this time, possibly for spawning. Therefore, reduction in river flow of the Flinders and Gilbert rivers may result in detectable changes in population size and/or dynamics.

Risk scores: Consequence **2**; Likelihood **3**. **Overall risk rating**: MODERATE

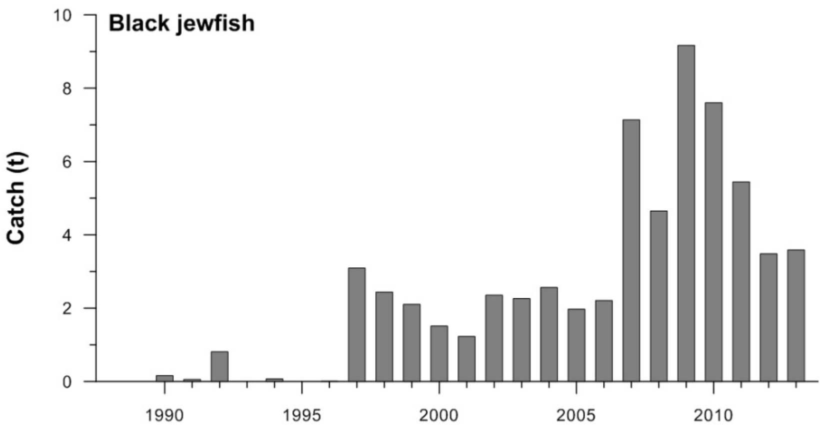

Figure 2.53. Annual catch of Black Jewfish (*Protonibea diacanthus*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

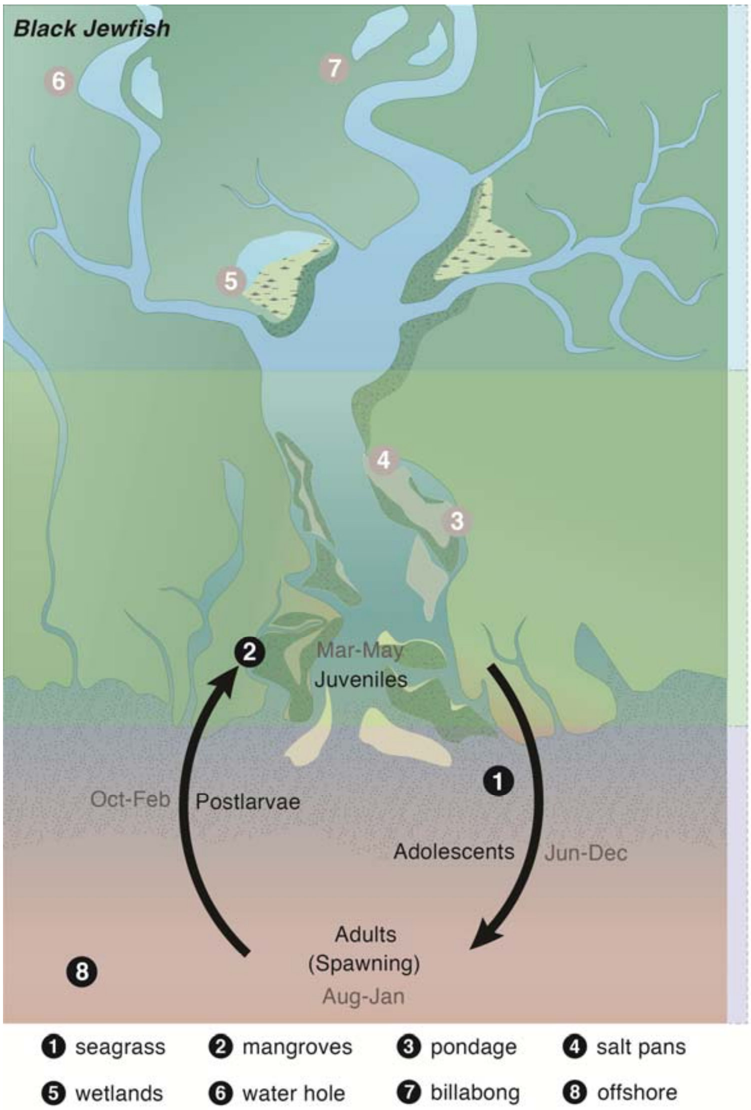

Figure 2.54. Conceptual model of the life history of Black Jewfish (*Protonibea diacanthus*) illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.20 Blue spot Trevally (*Caranx bucculentus*)

Blue spot Trevally (*Caranx bucculentus*) is a betho-pelagic species distributed throughout the Indo-West Pacific from Taiwan in the north to Gladstone, Australia in the south. It is commonly found in inshore shallow waters and are found in a variety of habitats such as sandy, muddy and seagrass habitats, over reefs, in estuaries, or large embayments. (Brewer et al., 1989; Brewer et al., 1994; Blaber et al., 1995).

It is a highly productive species, with Blaber et al. (1994a) estimating that its standing biomass was 2.6 kg ha^{-2} in the GoC and is the most abundant second order consumer. The species grows rapidly at 82 mm per year and lives for at least eight years (Brewer et al., 1994).

Blue spot Trevally matures early in life at around age 1 (~110 mm SL) and females produce up to 650,000 eggs per spawning. There is evidence from high gonadosomatic index values and year-round recruitment of juveniles to suggest the species spawns continuously throughout the year in the GoC. However, major peaks in spawning activity and recruitment occur in the lead up to the wet season in spring (Brewer et al., 1994), which may indicate river flows are a cue for spawning. The location(s) of spawning is not well understood, but it's likely that spawning takes place at multiple sites throughout the GoC. The sex ratio of Blue spot Trevally in the GoC is highly skewed towards males at small size classes, but towards females at larger sizes, indicating the possibility of protandous sex inversion.

Analyses of fish caught across the Gulf showed that including paenaid prawns form a major portion of blue spot Trevally diet (Brewer et al., 1989; 1991; Brewer et al., 1994). Blue spot Trevally is an important predator of paenaid prawns in the Gulf of Carpentaria with high blue spot abundances recorded from prawn trawling grounds, likely attracted by high prey availability (Brewer et al., 1989; Smith et al., 1992; Brewer et al., 1994).

Ecologically, Blue spot Trevally is one of the most important predators in coastal and estuarine habitats throughout the GoC. Its feeding ecology has been has studied extensively in and around the estuaries of the GoC (Brewer et al., 1989; Laprise and Blaber, 1992; Smith et al., 1992; Brewer et al., 1994; Salini et al., 1998) where they primarily consume small fish and benthic crustaceans, particularly penaeids. It has a high prey consumption rate of up to 7.3% body weight per day (Smith et al., 1992), thus having the ability to exert high predation pressure on lower trophic levels. In fact, Brewer et al. (1994) showed it is the most important predator of commercially important prawns in the GoC. Smith et al. (1992) quantified the predatory impact on commercially important prawns to be 11 g per day per kg of Blue spot Trevally.

Despites the high ecological importance of Blue spot Trevally, little is known of its population structure or movements. Of the 23 adult fish (540 to 651 mm FL) tagged in Albatross bay by the CSIRO (Griffiths, unpublished data), no recaptures have been recorded. Given its widespread distribution, high spawning frequency and broadcast spawning mode, it is probable that the species exists as a single stock throughout the GoC.

Blue spot Trevally is not considered to be of commercial importance to commercial, recreational or Indigenous fisheries. The species was not recorded in the 2010 recreational fishing survey or in commercial fishery logbooks, but would most likely be recorded under high-level aggregations of 'Trevallies'. Anecdotal accounts suggest that it is often caught by recreational fishers as adults around structure, but it is not taken as a food fish or as a primary sportfish.

Risk score justification

Blue spot Trevally is widespread in the GoC and uses a range of habitats, mostly in the nearshore region. The species is commonly found in estuaries but does not have a high dependence on this habitat to complete its life cycle (Figure 2.55). It is a highly productive species and there is little evidence for population subdivision. The species is known to rely heavily on commercially important prawns as prey.

Therefore, a reduction in river flows may reduce the available prey for this species, and also affect the possible cue for spawning.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

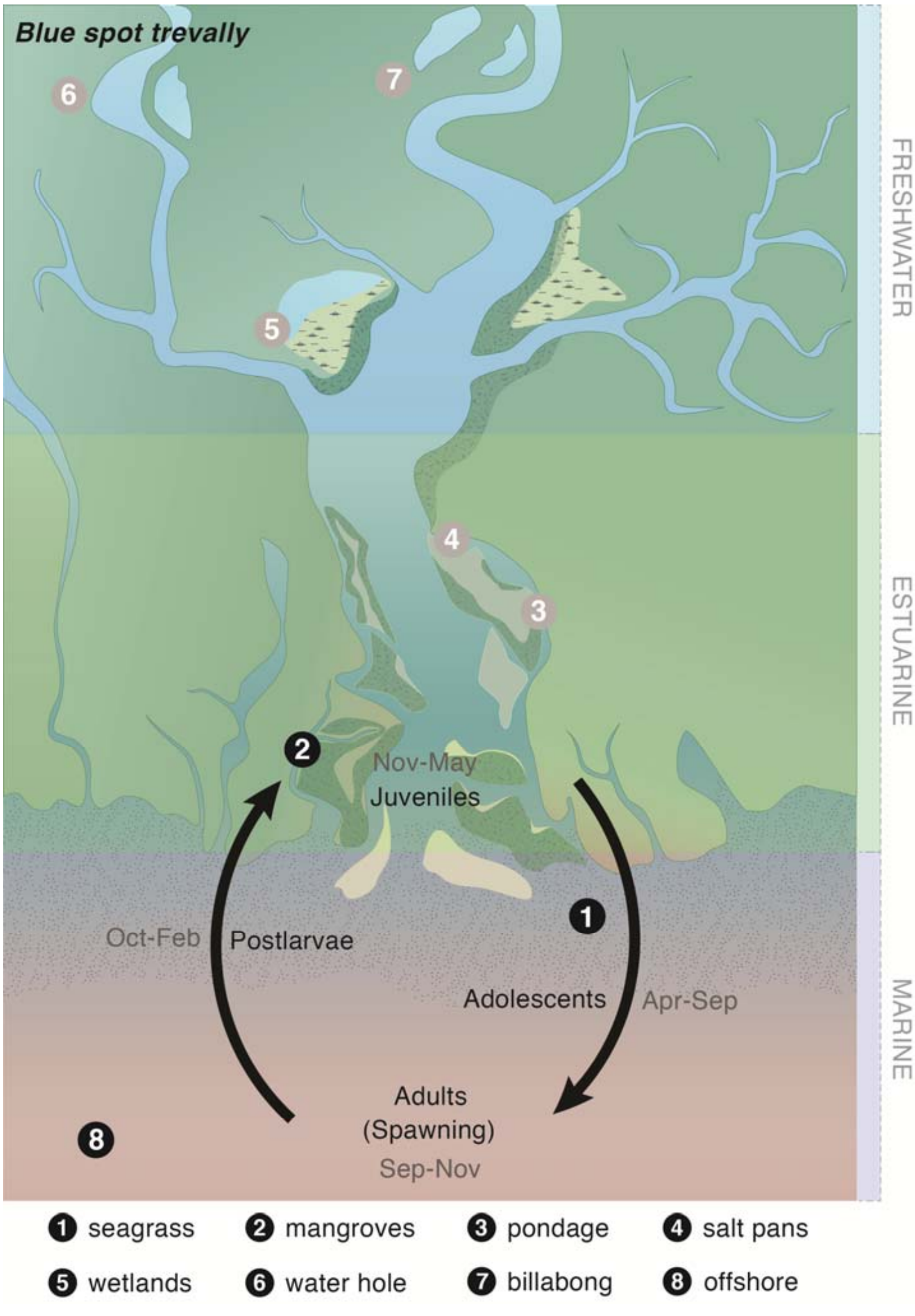

Figure 2.55. Conceptual model of the life history of Blue spot Trevally (*Caranx bucculentus*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.21 Blue Catfish (*Neoarius graeffei*)

Blue Catfish (*Neoarius graeffei*) is a benthic predator that is endemic to southern New Guinea and northern Australia, found distributed from the Ashburton River in Western Australia, to the Hunter River in New South Wales (Froese and Pauly, 2014). It prefers the soft-bottomed habitats within rivers, estuaries and coastal waters. Although the species can tolerate a wide range of salinities (9 to 36.0 PSU) and turbidity (0.8 to 19.0 NTU) in the GoC (Cyrus and Blaber, 1992), it generally prefers more turbid waters with higher salinities in the lower reaches of estuaries.

Little is known of the life history of Blue Catfish, with the exception of extensive research on its diet and dentition. This is a result of at least 13 closely related species of similar appearance occupying GoC estuaries, which can often only be identified by close inspection of the mouth gape and tooth plate arrangements (Blaber et al., 1994b).

Blue Catfish has been recorded to grow to a maximum size of 60 cm SL (Froese and Pauly, 2014), although its longevity is unknown. The only available information on the biology of Blue Catfish is from the southern end of its Australian distribution in the Clarence River, New South Wales. The species appears to mature reasonably early in life, with 50% of females being mature at less than half of its maximum length of around 280 mm SL (Rimmer, 1985a). Spawning takes place annually over a short period from November to December with rising water temperature and increased day length proposed to be the primary cue for spawning (Rimmer, 1985b). As with many species of catfish, Blue Catfish is a mouthbrooder. Females have low fecundity, producing between 40 to 122 oocytes that undergo buccal incubation by male fish following fertilisation for a period of 6 to 8 weeks, during which time the males cease to feed (Rimmer, 1985c).

In the GoC, Blue Catfish moves into seasonal waterways during the wet season to feed, and possibly to spawn. Catfish species in northern Australian waters feed on invertebrates including penaid prawns and small fish (Salini et al., 1990; Brewer et al., 1991; Brewer et al., 1995; Farmer and Wilson, 2011). Floodplains and seasonally-flowing creeks are important foraging areas for Blue Catfish, and floodplain food sources contribute substantially to its production (Jardine et al., 2012).

No studies exist on the genetic population structure or movements of Blue Catfish.

Blue Catfish is an important species to commercial, recreational and Indigenous fisheries in the GoC. Although commercial catch data are not available by fishery, most of the catch comes from the N3 fishery. Annual catches have varied substantially, ranging from no catch in 2013 to peaks of 11 t and 10 t in 1994 and 2004, respectively (Figure 2.56). Although not likely to be a target species, Blue Catfish (or 'Forktail catfish') was the fourth most abundant species caught by recreational fishers in the GoC during the 2010 statewide survey (Table 2.5). Of the estimated 21000 fish caught, 97% were released (QFISH online database).

Catfish are culturally important to Indigenous fishers (Jackson et al., 2014) as well as being a food fish, and were the ninth most abundant species caught (by number) in the Queensland component of the 2000/01 NRIFS. An estimated 21,494 fish were caught, constituting 3% of the total Indigenous catch (Table 2.6). Assuming a weight of 300g, the total catch of Catfish by the Indigenous sector in 2001 was approximately 6.5 t. A large proportion of this catch is likely to be Blue Catfish.

Risk score justification

A reduction in wet-season flow and floodplain inundation therefore reduced extent and connectivity of estuarine, fluvial and floodplains habitats (Figure 2.57). This may negatively impact local numbers of Blue Catfish by reducing access to prey and spawning habitats. Because Blue Catfish is a mouthbrooder and females have low fecundity, there is potential for the population to rapidly decline. However, this can also be an advantage where access to key habitats is restricted by flow as the male can regulate the physicochemical conditions to increase egg and larvae viability (Rimmer, 1985c).

Risk scores: Consequence **2**; Likelihood **3**. **Overall risk rating**: MODERATE

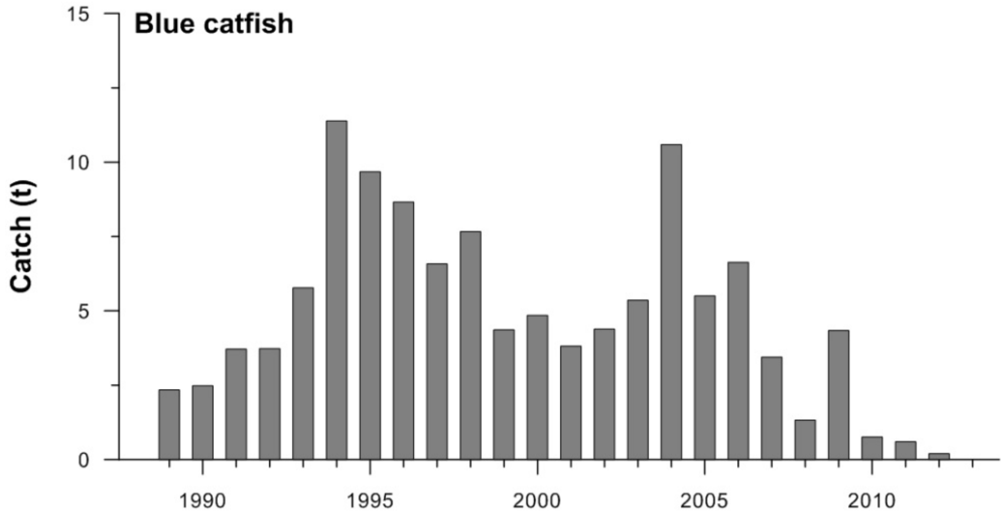

Figure 2.56. Annual catch of Blue Catfish (*Neoarius graeffei*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

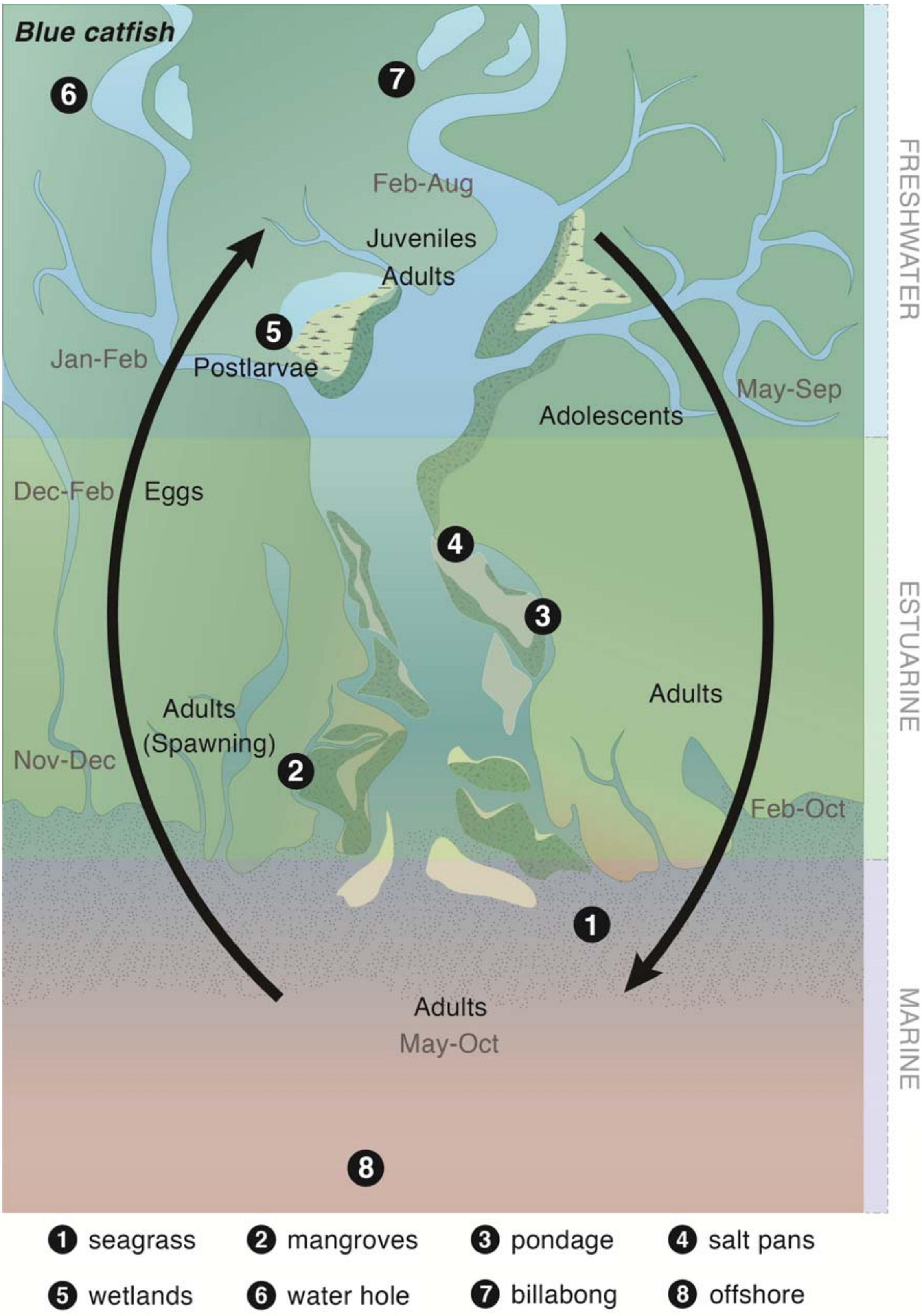

Figure 2.57. Conceptual model of the life history of Blue Catfish (*Neoarius graeffei*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.22 Mullet (*Liza vaigiensis* and *L. argentea*)

There are at least 20 species of mullet (family Mugillidae) recorded from GoC estuaries (Blaber et al., 1989; 1995). However, the species of highest importance to commercial, recreational and Indigenous fisheries, as well as the ecosystem overall, are the diamond scale mullet (*Liza vaigiensis*) and flat-tailed mullet (*Liza argentea*).

Mullet species occur in coastal, estuarine and fresh waters (Blaber et al., 1992; Blaber et al., 1995) and their lifecycle is similar to that of Barramundi (Halliday and Robins, 2005). Juveniles and adults are commonly found in estuarine and freshwater habitats (Grant and Spain, 1975; Robins and Ye, 2007). It is thought that freshwater is the preferred habitat for these species, with estuarine habitats used where access to freshwater is restricted (Halliday and Robins, 2005). Mullet tend to grow fastest during the wet season suggesting the influence of a seasonal increase in productivity of coastal waters (Grant and Spain, 1975).

Few quantitative studies have examined the biology of Mullet in Australia. There is some limited information on Flat-tailed mullet reproduction (Kendall and Gray, 2008) and growth (Grant and Spain, 1975; Kendall et al., 2009) from southeastern Australia. Flat-tail mullet grow to a maximum length of 45 cm TL (Froese and Pauly, 2014). In southeastern Australia, this species is slow-growing, reaching 50% of its maximum length by about age 5 to 8 years, and live to at least 17 years (Kendall et al., 2009). Histological analysis of gonads revealed that Flat-tail mullet mature early in life, with 50% of females mature by around age 2 to 3 years (207 mm TL) (Kendall and Gray, 2008).

Mullets spawn in marine waters in later summer when they aggregate in the lower reaches of estuaries before moving into coastal waters (Grant and Spain, 1975; Halliday and Robins, 2005). In Lake Maquarie, New South Wales, the spawning period extends from March to August, where on average, females produce 726,636 oocytes per spawning. Seasonal rainfall and flow likely influence the down-stream movement, either through responses to salinity changes or carried by high flows. Eggs and larvae require high salinity water. Post-larvae move back into estuaries from November onwards, potentially following salinity gradients (Halliday and Robins, 2005). A positive relationship was shown among catches of mullets and the extent and number of wetland patches; suggesting the extent and connectivity estuarine creeks and river channels are important to mullet production (Meynecke et al., 2008).

Mullet can be a reasonably important byproduct species to inshore commercial fisheries in the GoC. Although catch data from Queensland fisheries are not available by fishery, most of the commercial catch comes from the commercial N3 gillnet fishery. This catch is often used as an important bait species in the Mud Crab and recreational fisheries. Only a very small proportion of the commercial catch is separated into species. Diamond scale mullet is often separated, but most of the catch is recorded as 'Unspecified mullet' (Figure 2.58), which limits the ability to assess long-term catch trends by species. The annual commercial catch of Mullets has varied considerably, ranging from just 170kg in 2011 to 8 t in 2003 (Figure 2.58). In the past five years the average annual catch was 1.4 t.

Mullet are important bait species (live and dead) in the recreational fishery, however Mullets were not recorded to be caught by the charter fisher, and no catch estimate was available from the 2010 statewide recreational fishing survey.

The Indigenous fishery also regards Mullet as an important food species. The 2000/01 NRIFS recorded Mullets (unspecified) as being the fourth most important species caught (by number) in Queensland. An estimated 68,573 fish were caught, constituting 9.5% of the total Indigenous catch (Table 2.6). Assuming a weight of 200g, the total catch of Mullets by the Indigenous sector in 2001 was approximately 13.7 t.

Risk score justification

Mullets have a similar life history to Barramundi in that they spawn in the coastal regime and move back into estuaries where they use a range of estuarine and freshwater habitats (Figure 2.59). A reduction in flow volume and season may negatively impact Mullets through a reduction in the extent and connectivity of these estuarine and freshwater habitats, thus affecting growth and survival through a reduction in seasonal food accessibility. It may also negatively impact Mullet through a reduction in cues (e.g. flow, salinity) for movement out of low salinity waters for spawning. Therefore, a reduction in river flow of the Flinders and Gilbert rivers may result in detectable changes in population size and/or dynamics.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

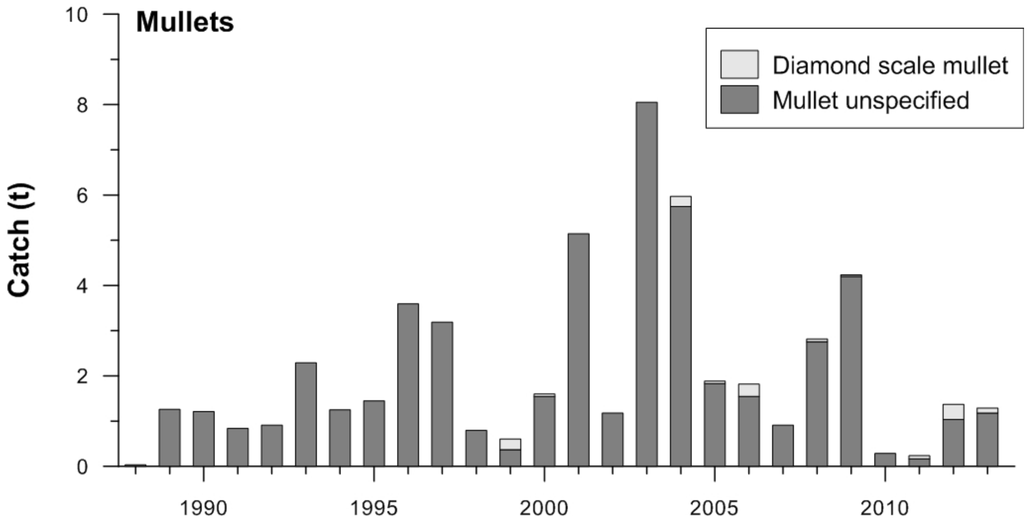

Figure 2.58. Annual catch of Mullet (primarily *Liza vaigiensis* and *L. argentea*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

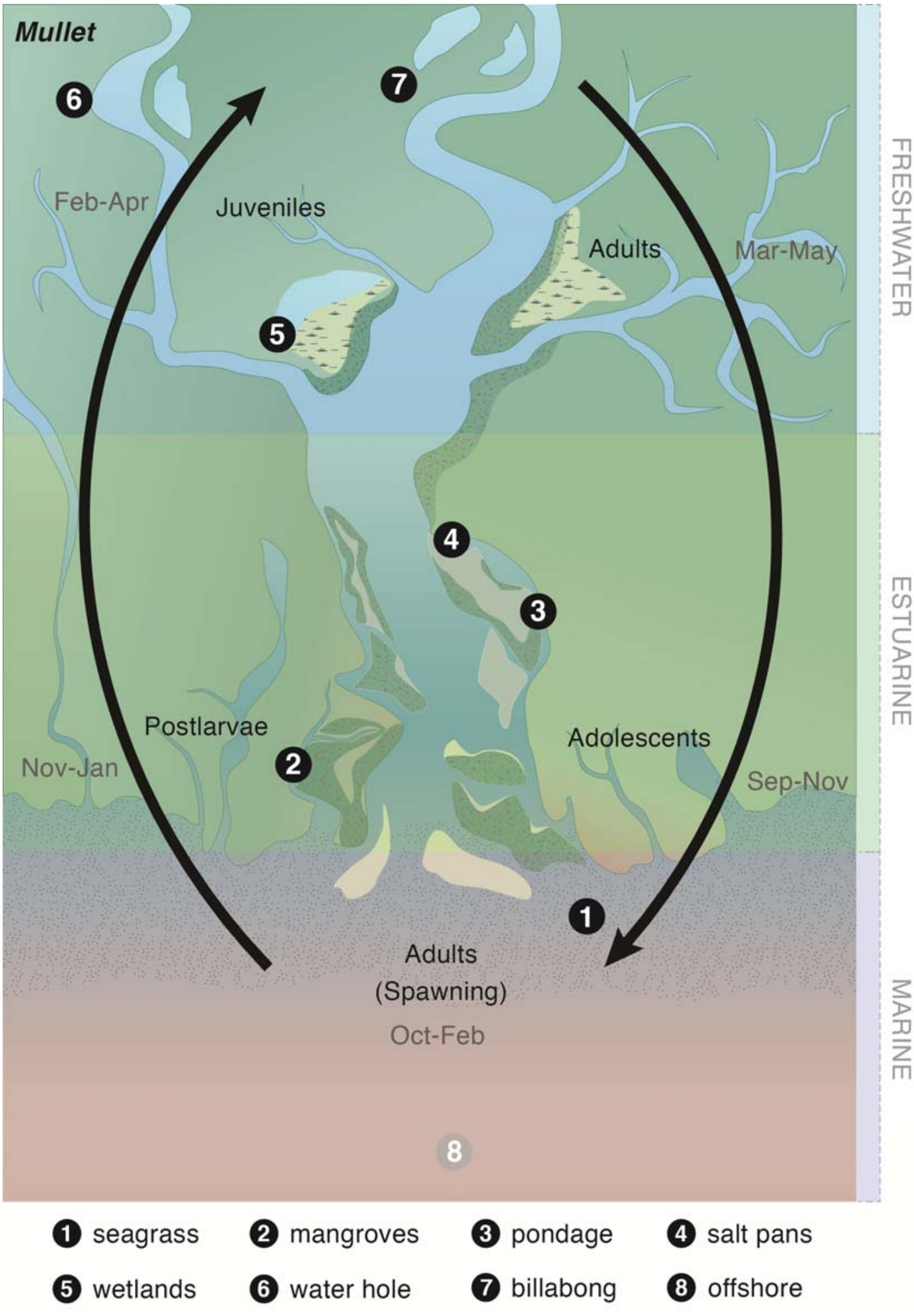

Figure 2.59. Conceptual model of the life history of Mullet (*Liza vaigiensis* and *L. argentea*), illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.23 Sooty Grunter (*Hephaestus fuliginosus*)

Sooty Grunter (*Hephaestus fuliginosus*) is a freshwater species that is endemic to northern Australia; its range extends from the Daly River, NT to Burdekin, Queensland (Froese and Binohlan, 2000). It is an iconic species to the Indigenous people of the GoC for food and recreational value (Jackson et al., 2014).

Sooty Grunter demonstrates a larval life history. It spawns in freshwater often associated with floodwaters. In the wet-dry tropics where river flows are intermittent, Sooty Grunter takes advantage of the wet-season floods and synchronises its spawning with the occurrence of floods. In perennial river systems, they spawn in the early summer, likely before eventual flooding. A key facet of its exposure to flood waters is that floods expand the habitat available for its larval and juvenile stages (Pusey et al., 2004).

The species spawns in aggregations in shallow, slack-water habitats adjacent to riffles/rapid habitats. These habitats are vulnerable to dewatering in the event of upstream regulation. Variation in Sooty Grunter recruitment strength is correlated with the extent of wet season flooding; probably due to the creation of new freshwater habitats and the stimulation of primary and secondary production in these supra-littoral habitats that support the juvenile phase of the species (Pusey et al., 2004). A second facet of spawning is that it occurs when water temperatures exceed 25°C; thus any impoundment infrastructure that impacts the natural thermal regime (stratified cold water releases) would disrupt the reproduction of Sooty Grunter. Some accounts suggest that on western Cape York Sooty Grunter migrates upstream to small tributaries to spawn; while in the Top End, it migrates out of the Arnhemland escarpment to the floodplain to spawn. Pusey et al. (2004) suggest that both these migrations are from dry-season remnant habitat to ephemeral wet-season habitats where Sooty Grunter spawns and the juveniles exploit the flood-stimulated habitat productivity.

Change in the extent of floodplain inundation due to water impoundment or diversion would negatively impact the creation of, and access to spawning and juvenile habitats of Sooty Grunter. Likewise, the restriction of baseflow due to impoundment would negate longstream connectivity that supports the early wet-season movement of adults from remnant waterholes to spawning habitat; and the return movement of juveniles and young adults from ephemeral habitats to deep riverine waters where they survive the dry season (Pusey et al., 2011a). Physical barriers to longstream movement would have the same effect.

Sooty Grunter is not caught commercially in GoC fisheries, but it is an important sportfish for recreational fishers and a culturally important food fish in the Indigenous fishery (Jackson et al., 2014). In the 2010 statewide recreational fishing survey, an estimated 19,000 fish were caught (Table 2.5). The survey estimated that 61% of recreationally-caught Sooty Grunter were released (QFISH online database). No species specific information was available from the 2000/01 NRFIS to provide an estimate of the catch in the Indigenous fishery.

Risk score justification

This endemic species will likely be affected by change in the extent of floodplain inundation due to water impoundment or diversion. Lower or no flows would negatively impact the creation of ephemeral wetlands which are likely spawning and juvenile habitats for these species. Likewise, the restriction of baseflow due to impoundment would negate longstream connectivity that supports the wet-season movement of adults from remnant waterholes in riverine habitats (Figure 2.60).

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

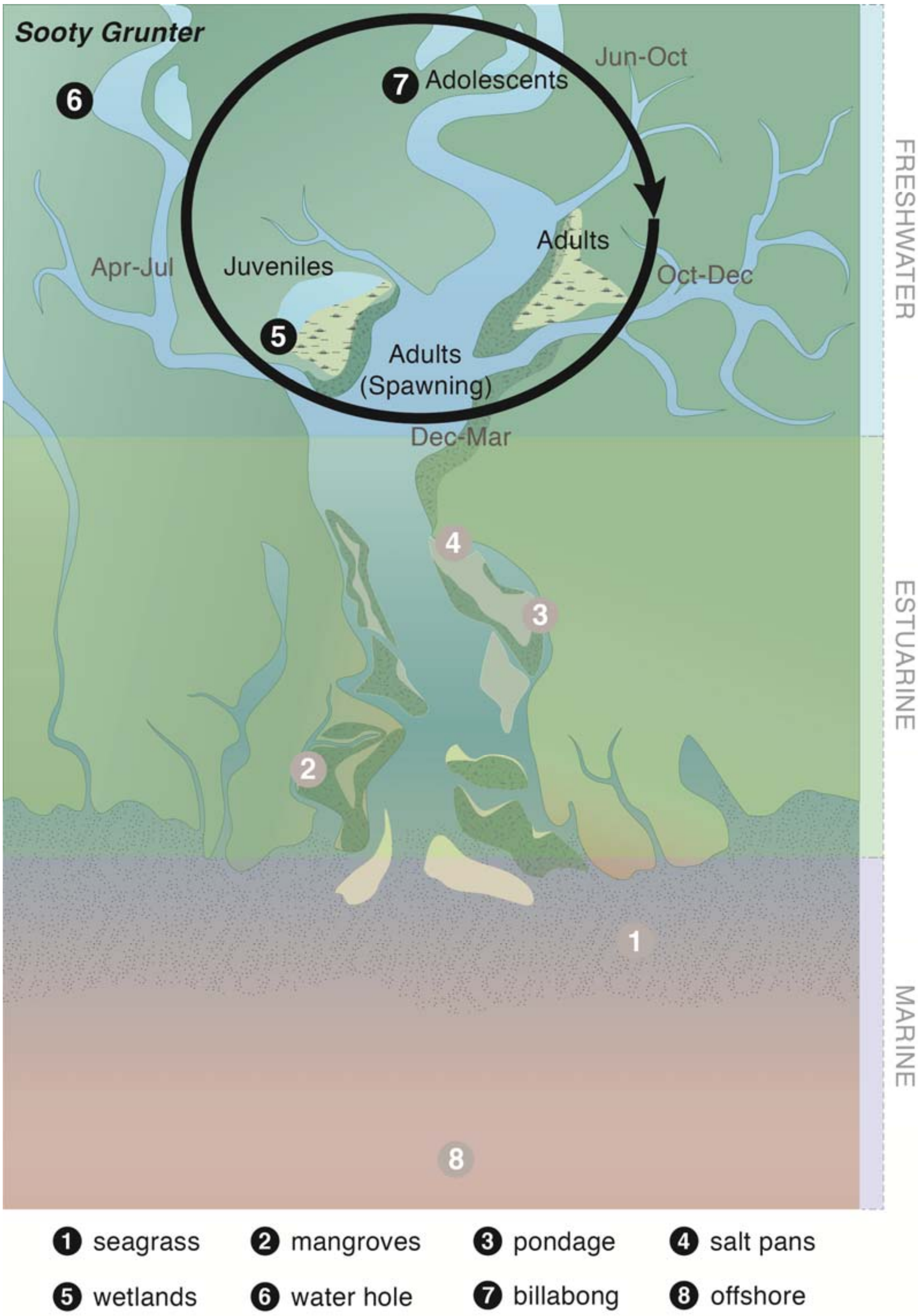

Figure 2.60. Conceptual model of the life history of Sooty Grunter (*Hephaestus fuliginosus*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.24 Slipper Lobsters (*Thenus parindicus* and *T. australiensis*)

Slipper lobsters species *Thenus parindicus* and *T. australiensis* (Scyllaridae) are common in the Gulf of Carpentaria and other Australian tropical coastal waters. They are byproduct species that support a major offshore GoC fishery and typify a life history strategy that would not be significantly impacted by interruptions to natural flow. Their entire life history occurs in offshore habitats (Milton et al., 2010). Jones (1993) suggests that *T. parindicus* is most abundant at depths of 20 to 30 m, whereas *T. australiensis* is most abundant at 40to 50 m depths. Thus, in the NPF *Thenus australiensis* only contributes a small proportion of the *Thenus* catch as its preferred habitat does not overlap greatly with the NPF.

The two species have similar seasonal spawning patterns, with the highest percent of berried females in October: 30 to 50% of the population of *T. parindicus* was berried from September to February, whereas 20 to 40% of *T. australiensis* was berried from August to January. Jones (2007) suggests that both species produce at least two egg clutches per year. Juvenile recruitment occurred from January to March and both species were found at most depths on the prawn grounds where they are caught as byproduct (Kenyon et al., 2011). Juveniles of both species (~20 to 30 mm CL) are common during the prawn surveys conducted in January each year, and sparse during similar surveys conducted in July. Large individuals of *T. parindicus* are common all year round, resulting in a strong bi-modal length frequency distribution for the species over summer (Kenyon et al., 2011). This trend is less evident for *T. australiensis* although the count for this species is much lower. The carapace length of mature females of *T. parindicus* ranged from 30.3 to 81.5 mm, and from 52.7 to 89.5 mm for *T. australiensis*. The estimated mean carapace length of females at sexual maturity (CL_{50}) was 52.0 ± 0.5 mm for *T. parindicus*, and 58.9 ± 0.5 mm for *T. australiensis* (Tonks et al., 2011).

Thenus parindicus matured at a smaller size and was less fecund than *T. australiensis*. The average fecundity of *T. parindicus* (40 to 70 mm CL) and *T. australiensis* (61 to 75 mm CL) is 11,350 ± 328 and 16,829 ± 2481 eggs per individual, respectively. The size at which 50% of females are mature (CL_{50}) suggests that current minimum legal size for *Thenus* is probably adequate for *T. parindicus*, but allows retention of immature *T. australiensis*.

Slipper Lobsters are only of importance to commercial fisheries in the GoC (primarily the NPF), as the depth of their primary habitat is generally beyond the limits of recreational and Indigenous fishers. The commercial catch of Slipper Lobsters is spread throughout the entire GoC fished area. The mean retained catch rate in most regions was <25 kg d^{-1}, but there are localised areas where >100 kg d^{-1} have been landed. These were mostly in the south eastern Gulf around Karumba (Zone 8) where the catch comprised almost exclusively *T. parindicus*. In Zone 8, NPF catches of Slipper Lobsters has varied significantly since the species began to be reported in 1998, ranging from 0.5 t in 2010 to 45 t in 2013 (Figure 2.61). The average annual catch in the past five years was 14 t.

Risk score justification

Reduced flows are unlikely to have a detectable effect on the populations of each species of Slipper lobster, as their life cycle (including spawning) is completed offshore, generally beyond the direct influence of the flood plume (Figure 2.62).

Risk scores: Consequence **0**; Likelihood **2**. **Overall risk rating**: NEGLIGIBLE

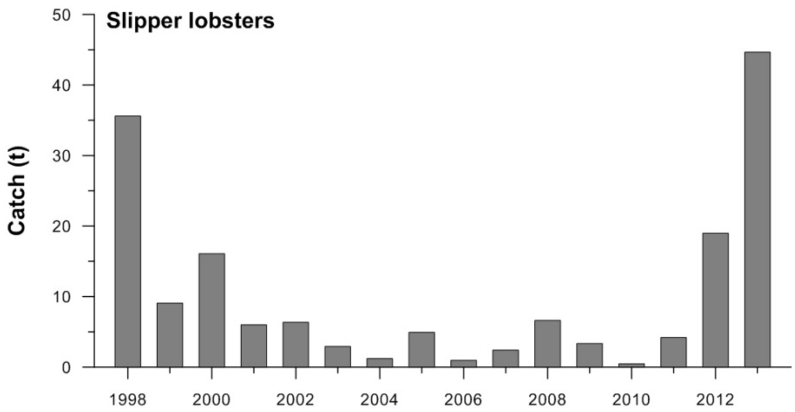

Figure 2.61. Annual catch of Slipper Lobsters (*Thenus parindicus* and *T. australiensis*) by the Northern Prawn Fishery in reporting Zone 8 in the Gulf of Carpentaria. Data supplied by the CSIRO.

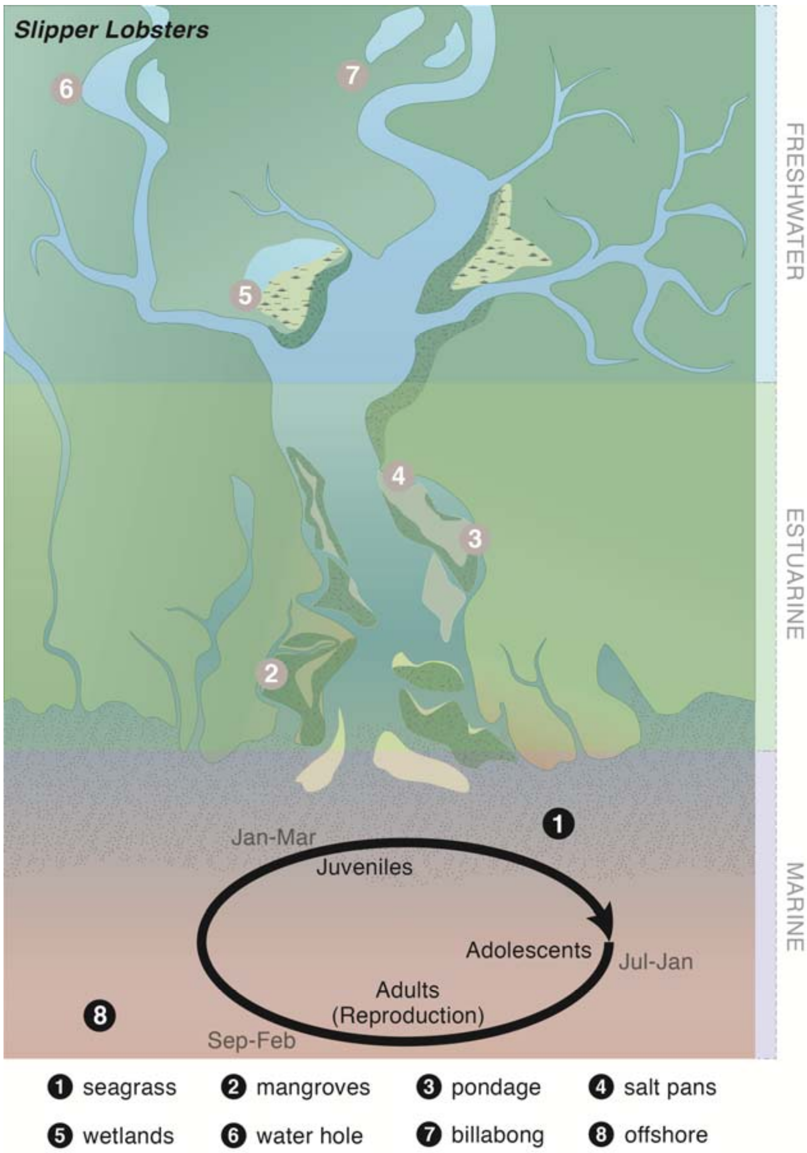

Figure 2.62. Conceptual model of the life history of Slipper Lobsters (*Thenus parindicus* and *T. australiensis*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.25 Cephalopods

Squid (Loliginidae) and cuttlefish (Sepiidae) are families of cephalopod (Phylum Mollusca; Class Cephalopoda) common in the GoC and other Australian tropical coastal waters. Most species are confined to offshore habitats (Milton et al. 2010) and occur throughout the prawn fishing grounds (Dunning et al., 1994) and would most likely not be significantly impacted by interruptions to natural river flows.

In the NPF, these are a byproduct species that support a major offshore GoC fishery. The squid catch in the NPF consists of several species: *Uroteuthis* sp. 1, *Uroteuthis* sp. 2, *Uroteuthis* sp. 3, *Uroteuthis* sp. 4, *Sepioteuthis lessoniana* and *Aestuariolus noctiluca*, but is dominated by *Uroteuthis* sp. 3 and *Uroteuthis* sp. 4. Likewise, the cuttlefish catch is comprised of five species (*Metasepia pfefferi*, *Sepia elliptica*, *S. papuensis, S. pharaonis* and *S. smithi*), with *S. elliptica* being the most abundant and widespread (Milton et al., 2010). These squid and cuttlefish are likely to be widespread Indo-Pacific tropical species. In the Gulf, squid are more abundant in shallower waters, especially the smaller species (*A. noctiluca*, *Uroteuthis* sp. 1 and *Uroteuthis* sp. 2). These species are also more abundant in areas with muddier substrates, whereas *Uroteuthis* sp. 4 is more abundant on sandier sediments (Milton et al., 2010). While the abundance of cuttlefish varies spatially, most are more abundant in deeper waters.

Squid and cuttlefish species are short lived animals (probably < 1 year) that spawn frequently throughout the year in relatively predictable locations in the southern GoC. Squid (*Uroteuthis* spp.) spawn with a greater intensity during May to November (the dry season). Mean gonadosomatic indices (GSI) were higher later in the dry season (August – October) for both *Uroteuthis* species and *Sepia smithi* and *S. papuensis*. The seasonal pattern of reproduction was less clear in the other two species of cuttlefish (*S. elliptica* and *S. pharaonis*), with a similar mean GSI throughout the year.

In the GoC, squid spawn in annual aggregations that seem to occur in the same area at the same time (from commercial logbook data). Some fishers return to these spawning areas each year and target the aggregations. Up to 200 t of squid (mostly *Uroteuthis* sp. 4) can be caught during a short period, suggesting that most of the animals in the aggregation are caught.

Each year, a large squid aggregation north-west of Mornington Island is regularly targeted by fishers. In May 2007 it comprised only *Uroteuthis* sp. 4. Female *Uroteuthis* sp. 4 in this sample were either in spawning condition or recently spent. This, along with squid egg clusters in the trawl nets, indicated that this was a spawning aggregation. Other similar large catches of squid were made near Groote Eylandt in several years (September to October); suggesting that these also were spawning aggregations of *Uroteuthis* sp. 4 or *Uroteuthis* sp. 3.

Cuttlefish are abundant in the southern and western parts of the GoC (from commercial logbook data). Catch rates of cuttlefish were lower than for squid, with the highest catch rates coming from the Vanderlins region. The year-round spawning pattern varied among cuttlefish species, with the greatest proportion of each species in spawning condition during the dry season throughout the deeper parts of the Gulf (25 to 40 m). Some species such as *S. smithi* and *S. pharaonis*, only spawn in the north-western GoC around Groote Eylandt. The shallow water adjacent to Karumba was the only region where *S. papuensis* and *S. pharaonis* did not spawn. The other two species, *S. elliptica* and *S. smithi*, spawned throughout the year in all regions (Milton et al., 2010).

Cuttlefish are less fecund than squid; the average fecundity of cuttlefish (66 to 190 mm mantle length (ML)) was 172 ± 18 to 343 ± 40 eggs per individual among four species, whereas fecundity for squid (59 to 222 mm ML) was 3341 ± 1001 and 4779 ± 677 between two species. There were considerable differences in the sizes at maturity for females among the six cephalopod species analysed. For example, *Uroteuthis* sp 3 had developed gonads when they were as small as 40 mm ML. This was a similar size to the smaller cuttlefish species, *S. elliptica* and *S. papuensis*. In contrast, *Sepia pharaonis*, *S. smithi* and *Uroteuthis* sp 4 appear to mature at larger sizes, 60 to 90 mm ML. Cuttlefish appear to compensate for their lower productivity by having larger eggs that are better protected in an egg case. They are also much longer-lived than squid and presumably spawn in successive years (Milton et al., 2010).

Squid are generally caught at higher rates in the shallower inshore areas of the Gulf. Catches for the smaller squid species, *Uroteuthi*s sp. 1, *Uroteuthis* sp. 2 and *A. noctiluca*, were highest in waters less than 10 m deep and where the substrate was predominately mud (Milton et al., 2010). The commercial catches for these three species appeared to be higher in areas where the sediment was mostly sand or mostly mud, compared to areas with a mixture of sand and mud. For the larger squid species, *Uroteuthis* sp. 3 appeared to be more abundant in deeper waters and *Uroteuthis* sp. 4 had higher catch rates in areas with sandier substrates.

The most common species of cuttlefish caught in the NPF prawn monitoring surveys (see Kenyon et al., 2011), *S. elliptica*, had a wide distribution from Groote Eylandt in the west to Weipa in the east (Milton et al., 2010). During 1990 – 1991 research surveys in the Gulf, this species was also the most widely distributed and abundant cuttlefish (Dunning et al., 1994). These surveys also showed the two other common cuttlefish species, *S. pharaonis* and *S. papuensis*, were most abundant in the south to south-east Gulf and east Gulf regions, respectively (Dunning et al., 1994). The impact of reduced flows on the population of squid and cuttlefish in the GoC would be limited to large-scale changes in the ecology and production of the Gulf ecosystem due to reduced nutrient transfer from terrigenous landscapes to the littoral zone during floodflows.

Cephalopods are important to the commercial NPF. They are likely to be opportunistically caught in small quantities by recreational and Indigenous fishers, but no data are available regarding catches in the region. Annual catches in Zone 8 of the NPF have been highly variable – due to the sporadic presence of aggregations – ranging between zero and 7 t in 2013 (Figure 2.63).

Risk score justification

Squid and Cuttlefish complete their life cycle (including spawning) in coastal and offshore waters of the GoC, where they exist as individual stocks (Figure 2.64). They are highly fecund, fast growing and short lived meaning they are highly resilient to changes in population reduction (e.g. fishing or environmental perturbations). Therefore, reduced flows is unlikely to have detectable effects on the populations of these species.

Risk scores: Consequence **0**; Likelihood **2**. **Overall risk rating**: NEGLIGIBLE

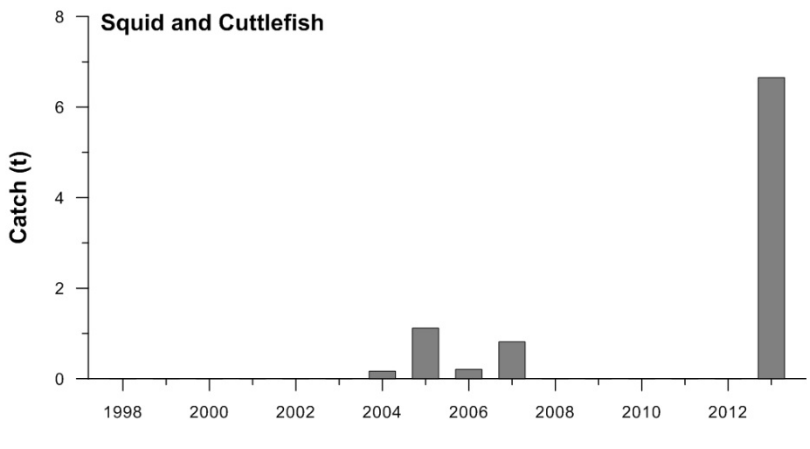

Figure 2.63. Annual catch of Squid (Loliginidae) and Cuttlefish (Sepiidae) by the Northern Prawn Fishery in reporting Zone 8 in the Gulf of Carpentaria. Data supplied by the CSIRO.

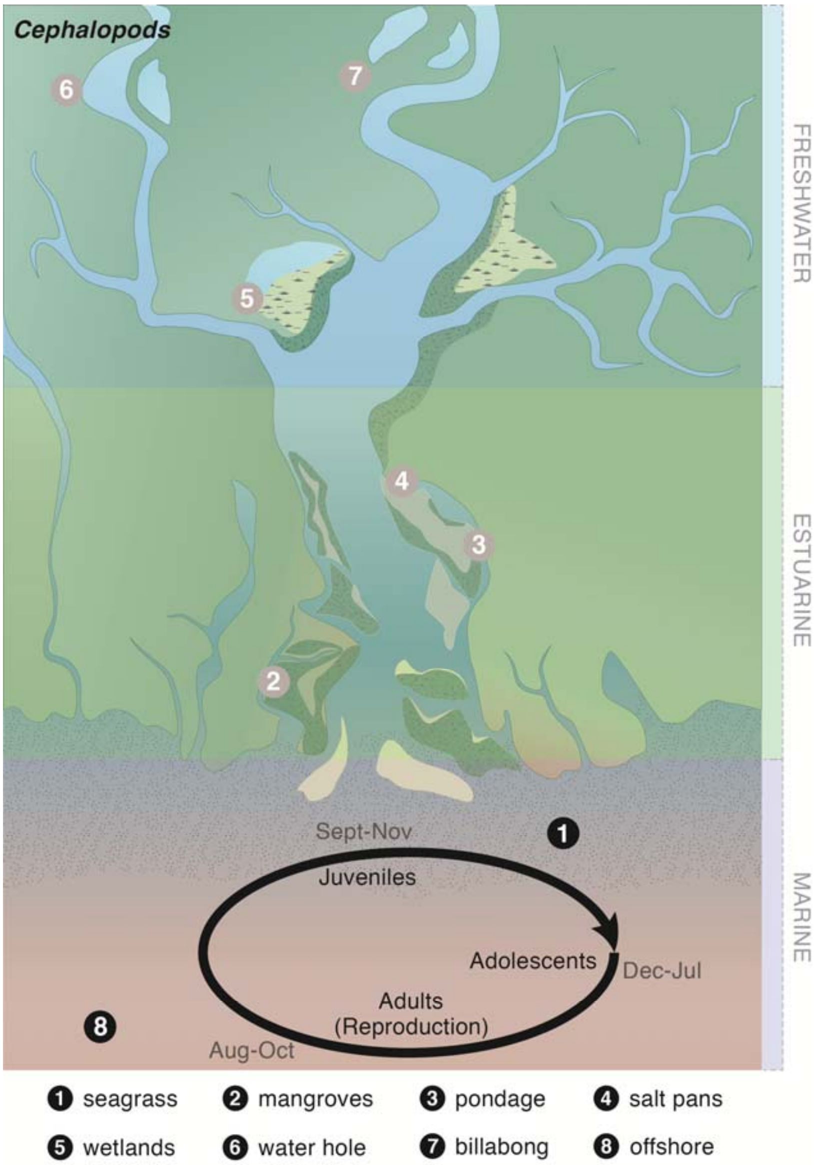

Figure 2.64. Conceptual model of the life history of Squid (Loliginidae) and Cuttlefish (Sepiidae), illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.26 Blue Swimmer Crab (*Portunis pelagicus*)

The Blue Swimmer Crab (*Portunis pelagicus*) is distributed throughout the tropical regions of the Indo-West Pacific and is common across northern Australia. It generally prefers shallow algal and seagrass habitats over sand and mud substrata (Williams, 1982; Edgar, 1990) located in turbid estuarine waters to clear offshore waters. However, the species has a preference for the high salinity of marine waters and even hypersaline estuaries of 30 to 40 PSU (Potter et al., 1983).

Blue Swimmer Crab is fast growing and most likely only live for up to three years (Sukumaran and Neelakantan, 1997). Females reach sexual maturity in their first year at about 60 mm CL in southern Australia and (Xiao and Kumar, 2004) and 86 to 98 mm CL in western Australia (De Lestang et al., 2003). Blue Swimmer Crab spawns from October to January in southern Australia (Kumar et al., 2003) and October to March in Western Australia (De Lestang et al., 2003). The sex ratio of the species in Western Australian estuaries is 1:1 (Potter and De Lestang, 2000). Juveniles recruit to inshore tidal flats, mangroves and the lower reaches of estuaries before moving to deeper offshore waters as adults(Svane and Cheshire, 2005).

Little is known of the horizontal movements of the species in Australian waters. A tagging study of 1003 crabs in Moreton Bay, Queensland showed that crabs moved only short distances (<10 km) (Potter et al., 1991). However, a genetic study of Blue Swimmer Crab revealed that widespread movement must occur with four subpopulations being identified within Australian waters; one occurring across northern Australia (Bryars and Adams, 1999).

Blue Swimmer Crab in Western Australian estuaries and coastal regions feed on a variety of benthic invertebrates, such as amphipods, polychaetes, bivalve molluscs, and also on teleosts such as gobies. In more tropical regions where prawn trawling occurs, a large part of its diet comprise discarded bycatch, primarily teleosts (Wassenberg and Hill, 1987).

In the GoC, Blue Swimmer Crab is an important byproduct species in the commercial Mud Crab fishery, and a food species for recreational and Indigenous fishers. The commercial catch in the GoC has been variable since 1988, with several years where no catch was recorded, to a peak of 3 t in 2007 (Figure 2.65). In the past five years the annual catch has averaged 1.3 t. Although Blue Swimmer Crabs is frequently caught by recreational fishers, it is not specifically recorded from the GoC in the 2010 statewide survey. The 2000/01 NRIFS revealed that the species was important to the Indigenous fishery where a large percentage of the estimated 2345 crabs recorded in the ‘Crabs (other)’ category (i.e. other than Mud Crab) was likely to be Blue Swimmer Crab.

Risk score justification
Blue Swimmer Crab is widespread in the GoC and uses a range of habitats, mostly in the nearshore regions. The species is commonly found in estuaries and mangroves as juveniles, where it spends the first year of life (Figure 2.66). It is a highly productive species and exists as a single population across northern Australia. A reduction in river flows may reduce the available prey and habitat for juveniles of this species, and may also affect the possible cues for spawning.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

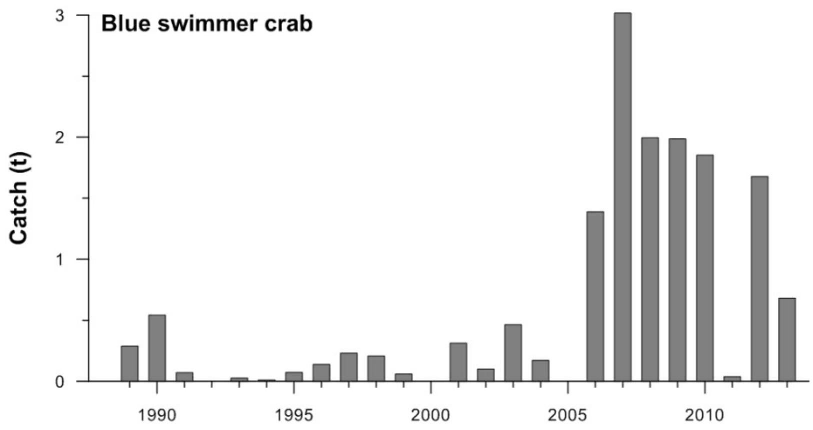

Figure 2.65. Annual catch of Blue Swimmer Crab (*Portunis pelagicus*) by Queensland's commercial fisheries (combined for all net, trawl, pot and line fisheries) in the Gulf of Carpentaria. Data supplied by Fisheries Queensland.

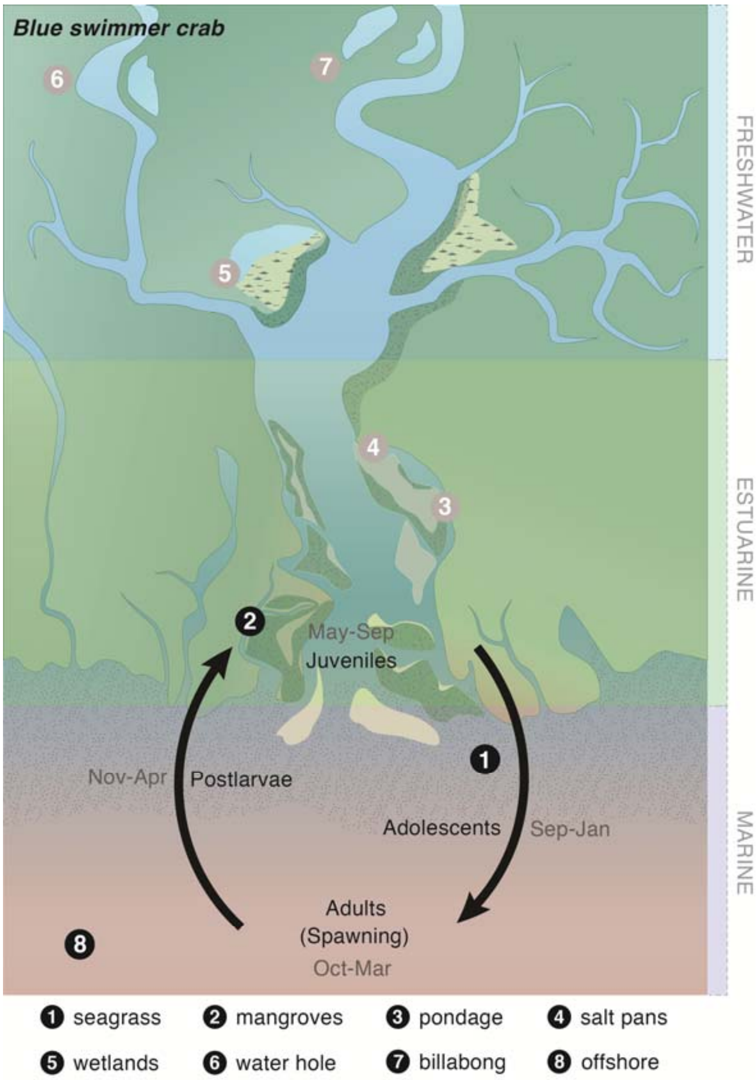

Figure 2.66. Conceptual model of the life history of Blue Swimmer Crab (*Portunis pelagicus*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.27 Sawtooth Barracuda (*Sphyraena putnamae*)

Sawtooth Barracuda (Sphyraena putnamae) is large pelagic predator that is widely distributed throughout the Indo-West Pacific (Froese and Pauly, 2014). It is primarily a reef-associated predator but is common in more turbid coastal waters, particularly in marine embayments and lagoons (Senou, 2001). It can tolerate a wide range of salinities (0.8 to 34.7 PSU) and turbidity (01.6 to 19.0 NTU) in the GoC (Cyrus and Blaber, 1992).

The species is a schooling fish that aggregates during the day but is more solitary during the night when it predominantly feeds. It was found to be the most important predator species (by weight) in the Norman River, with a mean density of 7.2 kg^{-1} hr^{-1} in gillnet sets. In the Norman River it was found to consume a wide range of prey, particularly fish, and to a lesser extent crustaceans, which is consistent with its diet elsewhere (Mohammadizadeh et al., 2010). However, it consumed significantly fewer commercially important prawns than other predators in the Norman River (Salini et al., 1998).

Although the species grows to a maximum size of 90 cm TL, it is present in GoC estuaries generally at half that size, indicating the possible nursery role of this habitat. Almost nothing is known of the life history of Sawtooth Barracuda including growth, reproduction and movement. However, several studies indicate that shallow, turbid water habitats such as mangroves and seagrass beds within the vicinity of estuaries are key nursery habitats for the juvenile phase of this species' life history (Blaber et al., 1995; Nagelkerken et al., 2000; Baker and Sheaves, 2005).

Sawtooth Barracuda is not considered to be important to commercial, recreational or Indigenous fisheries in the GoC, or to Australia more broadly. Many species of Sphyraenids are toxic to humans via ciguatera toxins (Lewis and Endean, 1984) and the bioaccumulation of heavy metals (Matta et al., 1999). Therefore, this species is most likely not consumed for these reasons.

Risk score justification

Sawtooth Barracuda is a highly important predatory species within the estuaries of the GoC. It uses estuaries and nearshore habitats (e.g. mangroves) as a nursery and feeding ground for juveniles and subadults (Figure 2.67). However, its extensive use of offshore reef habitats as adults for feeding and most likely spawning suggests that a reduction in river flows may not have a large impact on the population size or structure of this species.

Risk scores: Consequence **0**; Likelihood **3**. **Overall risk rating**: NEGLIGIBLE

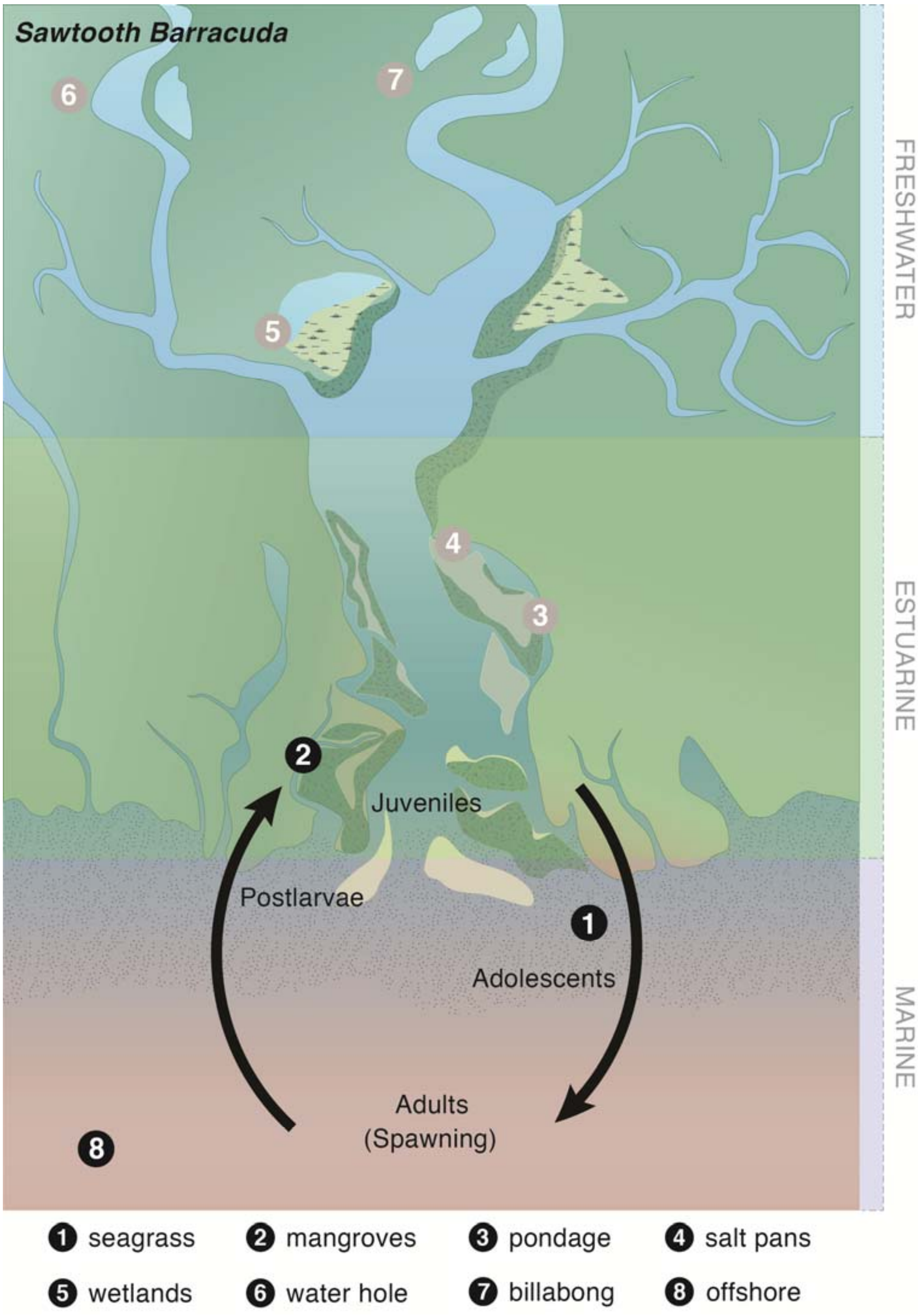

Figure 2.67. Conceptual model of the life history of Sawtooth Barracuda (*Sphyraena putnamae*), illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.28 Forage species

A very diverse range of fish species are important to the diets of predatory fish in tropical ecosystems such as the GoC (Blaber, 2000). Collectively, these ‘forage’ species help to maintain the integrity of marine ecosystems by supplying energy to species in higher trophic levels that support commercial, recreational and Indigenous fisheries, but also exert predatory pressure on lower trophic levels (Griffiths et al., 2013). In the estuaries of the south-eastern GoC, a vast number of species most likely contribute to this trophic regulation. However, for most species, there is little information on which to base a literature review. Here, a subset of species has been selected from the list of the most important prey species of Norman River predators (Table 2.2) to represent the subtly different life histories of forage species. These species are Bony Bream (*Nematalosa erebi* and *N. erebi*), Hamilton's thryssa (*Thryssa hamiltonii*), and Largescaled terapon (*Terapon theraps*). None of these species are of known importance to commercial, recreational or Indigenous fisheries.

Bony Bream is a benthopelagic clupeoid that represents an important forage species for predators throughout the length of the Norman River (Salini et al., 1998). This is because they can tolerate a wide range in salinity (0 to 30 PSU) and turbidity (0.8 to 21 NTU) (Cyrus and Blaber, 1992). However, these species are generally in highest abundance in the upper reaches of estuaries and the lower end of rivers where they are important in the diets of Barramundi and Sharks such as the Bull Shark (Blaber et al., 1989; Salini et al., 1990; 1992; Brewer et al., 1995). *Nematalosa erebi* has been documented to be highly fecund and spawn independent of flooding in the Murray River during summer (Puckridge and Walker, 1990). In other systems, abundance of Bony Bream has been documented to increase, where other native species have declined. It has been suggested (Puckridge and Walker, 1990) that this species is highly tolerant of low flow environments and may rapidly recover after significant perturbations. As a result the risk scores are: Consequence **0**; Likelihood **3**. **Overall risk rating**: NEGLIGIBLE

Hamilton's Thryssa is a pelagic schooling species that is widely distributed through the Indo-West Pacific, primarily occupying neritic waters. It is abundant in nearshore habitats and estuaries of relatively high turbidity (2.4 to 9.2 NTU), but it is not tolerant of low salinities and prefers a salinity range of 17.2 to 35.2 PSU that generally occurs in the lower estuarine and marine waters (Cyrus and Blaber, 1992). It primarily feeds on planktonic prey such as copepods, amphipods, and larval fish and prawns (Hajisamae et al., 2003). Little biological information is available for the species from Australian waters, but a study in the Arabian Gulf showed that the species is short lived (1-2 years), matures early in life and is a highly fecund broadcast spawner. Spawning takes place in the coastal regime over a long period from December to April (Hussain and Ali, 1987). Hamilton's Thryssa and ecologically similar species are therefore unlikely to be significantly impacted by reduced flows in the GoC. As a result the risk scores are: Consequence **0**; Likelihood **3**. **Overall risk rating**: NEGLIGIBLE

The Largescale Terapon is a small (maximum size 30 cm SL) demersal schooling species that is widely distributed through the Indo-West Pacific (Froese and Pauly, 2014). It is highly abundant in a wide range of habitats in the GoC from the lower reaches of estuaries, mangroves, reefs and offshore prawn trawl grounds (Blaber et al., 1989; Blaber et al., 1994a; Salini et al., 1998) where it prefers reasonably high salinity (12 to 36.1 PSU) (Cyrus and Blaber, 1992). In the GoC, it is primarily a benthic feeder, consuming a range of small fish, crustaceans and polychaetes (Dell et al., 2013). There have been no published biological studies on the Largescale Terapon, or other species in the *Terapon* genus. However, based on unpublished data from over 30 years of bycatch studies by the CSIRO, the species does not appear to have changed in abundance over time, indicating it is likely to be a highly productive and fecund species in the GoC. Combining this with evidence that the species prefers more coastal habitats of high salinity means that reduced river flows are unlikely to significantly affect the population of this, or ecologically similar species. As a result the risk scores are: Consequence **0**; Likelihood **3**. **Overall risk rating**: NEGLIGIBLE

8.29 Plankton

Plankton are the generally microscopic organisms that inhabit the oceans and are unable to swim against currents. The plankton includes autotrophic species – photosynthetic phytoplankton – and heterotrophic species – consumers known as zooplankton. The zooplankton can be further differentiated into the holoplankton, which are animals that spend their entire life cycle as plankton, and meroplankton, which spend only a portion of their life cycle as members of the plankton. In this review, a focus is made on available information on the plankton in the GoC.

There have been only a small number of studies investigating plankton in the GoC, most of which have primarily considered biomass measurements (Markina, 1972; Motoda et al., 1978; Rothlisberg and Jackson, 1982), however, Othman et al. (1990) undertook a detailed study on the copepods in the GoC.

Plankton biomass in GoC appears to be high in comparison to other areas around Australia. Mean biomass values of the GoC are some of the highest values seen around the Pacific Ocean perimeter, including the Eastern Tropical Pacific, fringes of the upwelling zones of North and South America, and some areas of equatorial convergence. Further standing stocks compare to other coastal areas supporting important fisheries of the west coast of North America, the eastern North Atlantic Ocean and some European waters (Rothlisberg and Jackson, 1982).

There are marked fluctuations in biomass and maxima appear in different parts of the Gulf at different times, although most often near the coast. For coastal plankton, there are higher abundances occur during the summer wet season, with the peak zooplankton biomasses recorded in October (Markina, 1972) and November (Rothlisberg and Jackson, 1982). Copepods and chaetognaths are abundant at most times when zooplankton biomass is high, but salps, siphophores, pteropods and other gelatinous zooplankton are also abundant at times. Groups that dominate the phytoplankton biomass include Nitzchia, Rhizosolenia, Coscinodiscus, Ceratium and chain-forming centric diatoms. The timing of the peaks in these coastal plankton might be attributed to nutrient input from terrestrial runoff during the wet season (Rothlisberg and Jackson, 1982).

Small neritic zooplankton, meroplankton and phytoplankton are most abundant in coastal stations in the GoC and this is probably best demonstrated by the copepod assemblage, which is the only group which has been examined in detail. The composition of this group is dominated by species characterised as being neritic (Othman et al., 1990). Although 88 of the 102 species of copepods identified by Othman et al. (1990) have also been described from South-east Asian waters, 95 copepod species had not been previously recorded in the GoC. Most of these species are mesopelagic or bathypelagic. Epipelagic copepod species are also absent from the Gulf, but there are a small number of copepod species that occur in the Gulf which could be considered oceanic.

More recent work has shown that there are three copepod assemblages in the GoC: viz. coastal, oceanic and transitional assemblages. During the wet season, the coastal assemblage pushes the transitional and oceanic assemblages further offshore. There was high coherence among the inshore assemblages throughout the Gulf, in terms of their movement offshore during the wet season. There was a more oceanic assemblage present during the dry season, when the influence of freshwater runoff from the rivers has diminished.

The copepod species in the GoC are mainly tropical/subtropical neritic species, but there are about 40 cosmopolitan species and 17 species that extend in temperate waters. Most are Pacific and Indian Ocean species but there are a small number of Atlantic species. Thirteen species that Othman identified had not been described before, all of these were neritic and it is likely these species are endemic to the GoC (Othman et al., 1990).

Risk score justification

The plankton community in the GoC is dominated by neritic species except for the very offshore region, and biomass peaks correspond to the wet season when out flows from the rivers are greatest (Figure 2.68). These outflows bring nutrient inputs from terrestrial sources at a time of year when environmental temperatures are high. In this shallow well-mixed embayment, turnover is likely to be fast, which can sustain high secondary productivity and in turn could sustain large vertebrate and invertebrate fisheries. Reduced flow rates are likely to greatly reduce the nutrient input, and therefore the productivity of the waters, which is likely to directly impact both the plankton, and larger animals which feed upon them.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

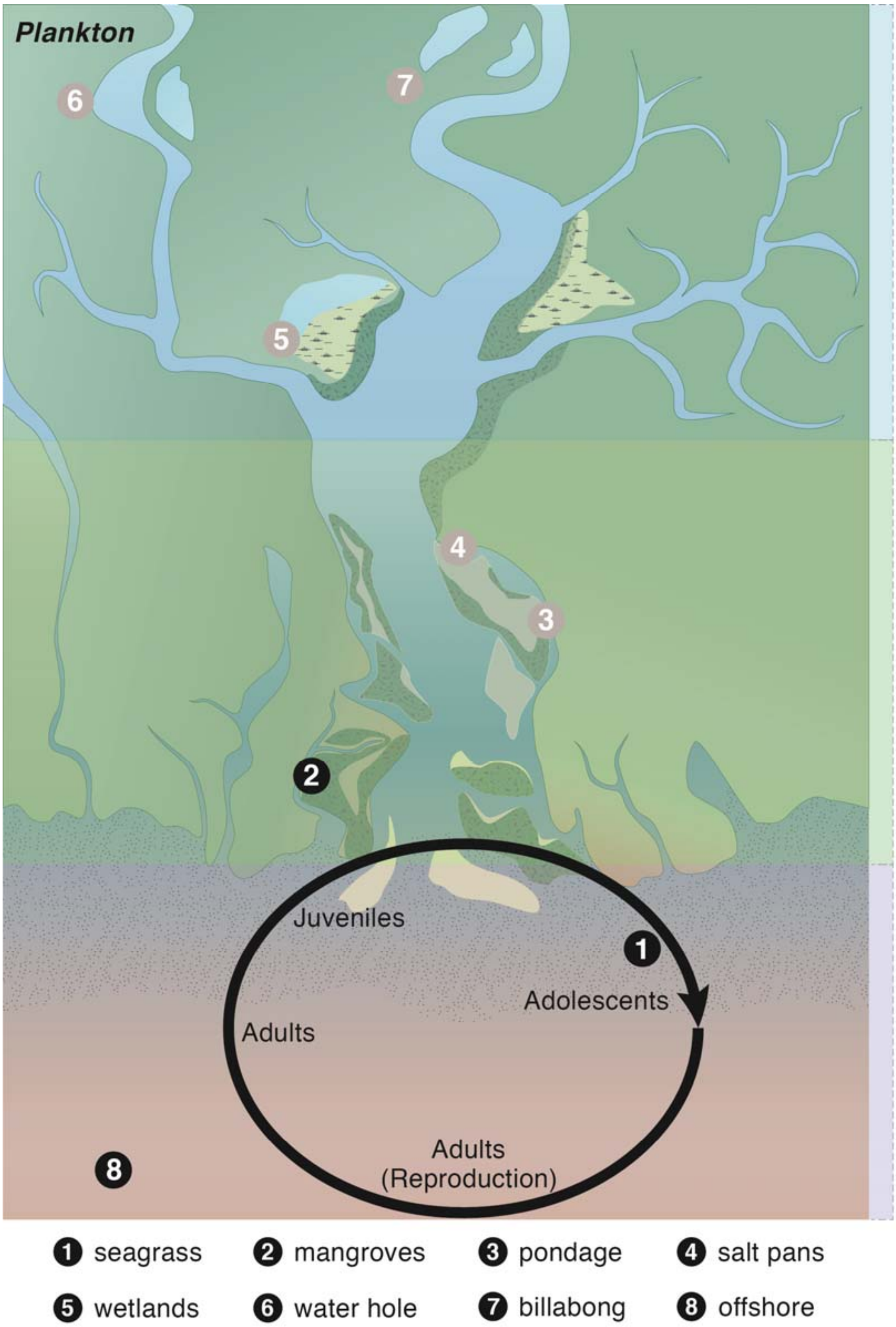

Figure 2.68. Conceptual model of the life history of Plankton, illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.30 Sea snakes

Twenty five seasnake, mudsnake and filesnake species occur across tropical northern Australia and none are considered endangered, threatened or vulnerable. However, all species of true seasnake (21 Hydrophiidae) are listed under the EPBC Act and several are endemic to Australia and southern Papua New Guinea. Two species of file snake (family Acrochordidae) inhabit tropical Australia, though only the little file snake (*Acrochordus granulatus*) lives in the marine environment (coastal rivers, estuaries and open sea). Three species of mudsnake (Colubridae) are found in the region. Until recently, it was considered that two species of true seasnake were found in coastal and estuarine habitats (*Hydrelaps darwiniensis*, *Parahydrophis mertoni*), together with mudsnakes (*Cerberus australis*, *Fordonia leucobalia*, *Myron richardsonii*) and the filesnake. *Cerberus australis* and *M. richardsonii* are endemic to Australia.

Fry et al. (2001) studied sea snakes in the NPF from both commercial vessel records and research surveys. In the NPF, sea snakes have been well surveyed as they are encountered as bycatch (Redfield et al., 1978; Wassenberg et al., 1994; Ward, 1996; 2000). Fry et al. (2001) found 17 species to be commonly caught in the offshore regions of the GoC. *Hydrophis ornatus* and *Disteira major* were the most abundant species caught commercially (73%); whereas *H. elegans* (30.8%) and *D. major* (27.8%) were most abundant in research surveys (Fry et al., 2001).

Many sea snake species are specialist feeders. The diet of nine species consists of only one to four prey species (Fry et al., 2001). More than 90% of the prey of *H. elegans*, *D. kingii* and *D. major* consisted of a single fish species. *Lapemis curtus* preyed on more than 20 fish and mollusc species.

All female *A. peronii*, *D. kingii*, *D. major*, *H. elegans*, *H. ornatus*, *L. curtus* were carrying full-term embryos between January to March and likely birthed between March and June (Fry et al., 2001). Female *A. eydouxii* carried full-term embryos later in the year; from May to August in offshore habitats. Many sea snakes had a clutch size smaller than ten animals (Fry et al., 2001).

Few juvenile sea snakes are caught in NPF trawl nets, perhaps because of their small size relative to the mesh size of the nets. However, survey and anecdotal information suggests that the juveniles of many of the offshore species may reside in estuarine habitats (Voris and Jayne 1979; Wassenberg et al. 1994; G. Fry CSIRO pers. comms.). A night-time survey in the Embley and Mission River estuaries recorded 15 species of seasnake, most of which were also common offshore (Table 2.14). The occurrence of over 50% of juvenile individual sea snakes among 4 of the 15 species collected highlighted the importance of estuarine nursery habitat for sea snakes.

Recent surveys in the GoC have used molecular analyses to identify new species of common genera in GoC estuaries (*Aipysurus* and *Hydrophis*) and to revise current species (Sanders et al., 2012; Ukuwela et al., 2012). Some species have not previously been encountered due to their estuarine habitat that has been poorly surveyed for sea snakes (Ukuwela et al., 2012). These discoveries suggest there may be known species or new species inhabiting poorly surveyed estuaries in the south-eastern GoC.

Risk score justification

Some species of sea snake have a high reliance upon estuaries during part of their life cycle, and some appear to use estuaries in the GoC for their entire lives (Figure 2.69). These EPBC listed reptiles are little studied in estuarine habitats, so the extent of their distribution into estuarine and possibly freshwaters and the effects of altered flow reigmes is poorly understood. Both the water characteristics of seasnake habitat and that of their prey may be impacted by reduced flow or change in the timing of flows. Juvenile sea snakes are present in the estuaries during the late wet season to early dry season (Voris and Jayne, 1979; Fry et al., 2001), so they would be exposed to annual flow events. Furthermore, their low fecundity and reproductive strategy of parental care are factors that may result in rapid declines in population size if flow regimes are unfavourable for rearing their young. Therefore, reduced flows may have a significant effect on recruitment levels or the capacity of the population to increase.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

Table 2.14. Habitat use and diet of sea snakes from the tropical Gulf of Carpentaria and Top End regions. (From Guinea at al. 2004) Estuarine sea snakes surveyed (and % juveniles when ≥15 sea snakes caught) by Porter et al. (1997) and Ukuwela et al. (2012) are marked *. Species endemic to Australia and Papua New Guinea are marked '*e*'. Recently described species are included #.

Species name	Habitat	Depth	Food
Family Acrochordidae			
Acrochordus granulatus *	Intertidal mud flats and mangrove	< 20 m (15% juveniles)	Small fish and crabs
Family Colubridae			
Cerberus australis e	Estuaries and mangroves	< 10 m	Small fish
Fordonia leucobalia	Mudflats and mangroves	< 10 m	Crabs
Myron richardsonii e	Estuaries and mangroves	< 10 m	Small fish
Family Hydrophiidae			
Acalyptophis peronii *	Sandy substrates	< 20 m (15% juveniles)	Gobies
Aipysurus duboisii *	Coral reefs	<50 m	Fish in general
Aipysurus eydouxii *	Turbid waters	<50 m (90% juveniles)	Fish eggs
Aipysurus laevis -	Coral reefs	< 30 m	Fish in general
Astrotia stokesii *	Turbid and clear waters	<30 m (100% juveniles)	Scorpion fish
Disteira kingii *	Various habitats	<100 m	Fish
Disteira major *	Turbid waters	<100 m	Fish
Enhydrina schistosa *	Bays and estuaries	<10 m (50% juveniles)	Catfish
*Hydrelaps darwiniensis * e*	Mangroves and mudflats	<10 m	Gobies
Hydrophis caerulescens	Mud substrates	< 20 m	Eels and gobies
Hydrophis coggeri *	Sand around coral reefs	< 50 m	Eels
Hydrophis czeblukovi e	Deep water	< 50 m	Eels
*Hydrophis elegans * e*	Turbid reef waters	< 50 m (5% juveniles)	Eels
*Hydrophis donaldi * e #*	Estuaries		New species
Hydrophis inornatus *	-	-	
Hydrophis macdowelli	Turbid estuaries	< 30 m	-
Hydrophis ornatus *	Eurytopic	< 50 m	Fish
*Hydrophis pacificus * e*	-	< 50 m	-
Hydrophis vorisi e	-	< 50 m	-
Lapemis curtus *	Eurytopic	< 50 m (85% juveniles)	Fish in general
Parahydrophis mertoni e	Mudflats and mangrove channels	< 10 m	-
Pelamis platurus	Open water	Any depth	Pelagic fish

Lapemis curtus is a revised species name for *L. hardwickii*.

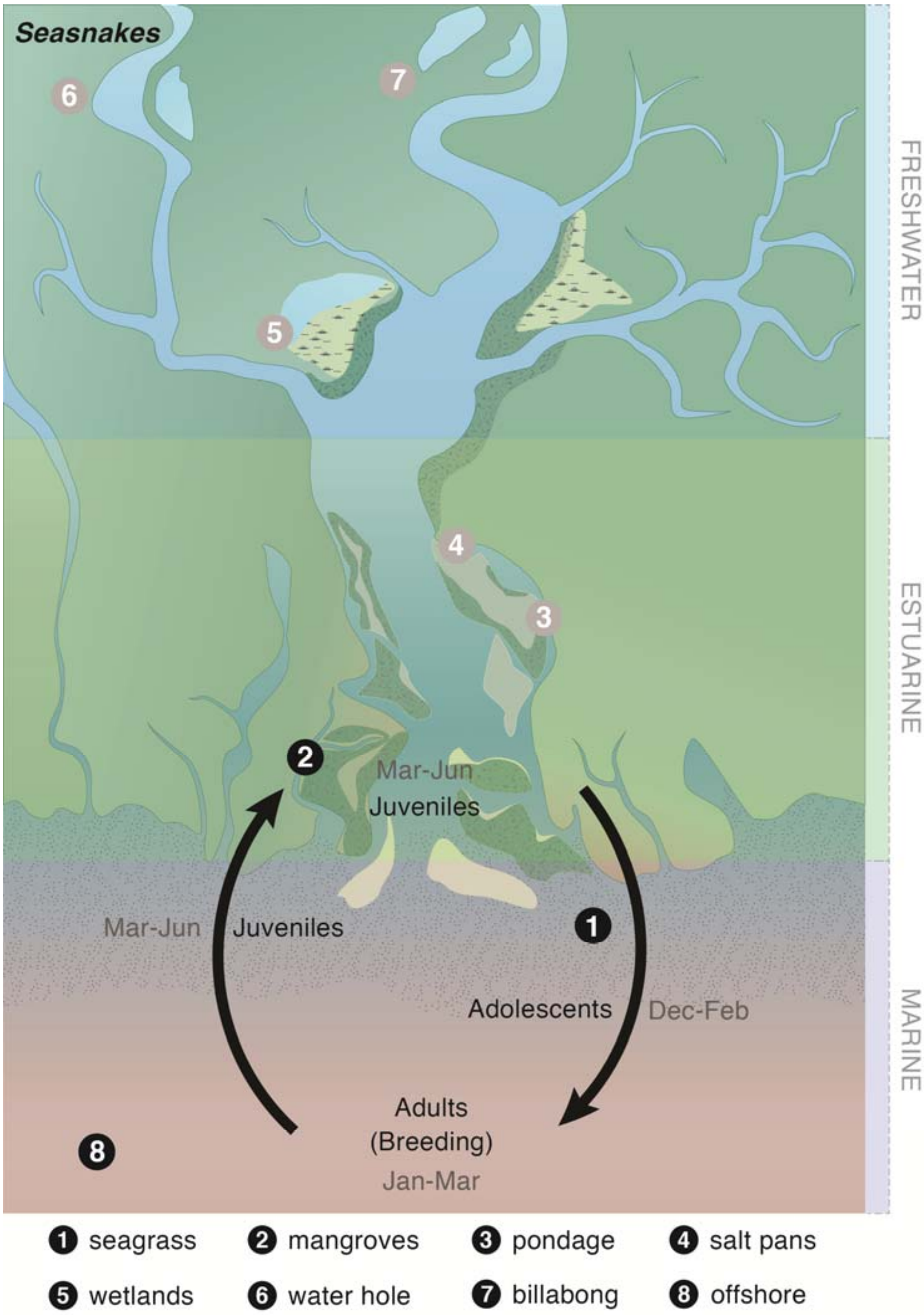

Figure 2.69. Conceptual model of the life history of Sea snakes, illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.31 Marine Turtles

There are six species of Marine Turtle that nest and forage in Australian waters: Flatback (*Natator depressus*), Green (*Chelonia mydas*), Loggerhead (*Caretta caretta*), Olive Ridley (*Lepidochelys olivacea*), Hawksbill (*Eretmochelys imbricata*), and Leatherback (*Dermochelys coriacea*) turtle. Turtles are important to Indigenous communities. Marine Turtles are very long lived and use different habitats during different life stages. Females gather near nesting beaches and lay eggs in nests high on the shore. Hatchling turtles disperse on the ocean currents, often for many years, before joining adults on foraging grounds in coastal waters. Large juvenile and adult Hawksbill, Loggerhead, Flatback and Green turtles can spend most of their time in foraging habitats. In the Australasian region, Flatbacks feed on soft-bodied invertebrates in often turbid waters (Limpus et al., 1983), Hawksbills feed on invertebrates on coral reefs and Loggerheads feed on invertebrates in coastal waters and estuaries. In contrast, Green turtles are primarily herbivorous feeding on mostly seagrass and algae (Garnett et al., 1985; Brand-Gardner et al., 1999; Andre et al., 2005; Fuentes et al., 2006). The two species of most relevance for this assessment are the Green and Flatback turtles.

The Flatback Turtle is only found in neritic waters of northern Australia, eastern Indonesia and southern Papua New Guinea and is considered endemic to Australia as it nests only on Australian beaches (Walker and Parmenter, 1990; Limpus, 2008). In the GoC, Flatback Turtles breed year round with a peak during the winter months (Limpus, 2008). There are four major nesting areas across northern Australia, representing four genetic stocks. The GoC and Torres Strait populations represent one stock. There are seven regional populations of Green Turtles that nest in different areas; with the GoC representing one population. Nesting occurs in the GoC year round, with a peak during the winter months. Green Turtles, nesting in the GoC, show restricted movements, nesting and foraging areas contained entirely within the GoC (Kennett et al., 2004).

The Green Turtle may be impacted by degradation of seagrass beds, through large flood events for example. This can lead to depressed growth rates of juveniles, fewer breeding adults, and increased mortalities (Chaloupka et al., 2004; Limpus et al., 2005). Flood events also wash nutrients from the land which can lead to harmful algal blooms in coastal waters with long-term detrimental impacts on the health of turtles (Arthur et al., 2008).

Catching Marine Turtles is prohibited in commercial and recreational fisheries, although they are caught as a bycatch in trawl and pelagic longline fisheries throughout their distribution (Lewison et al., 2004). However, changes in fishing practices have significantly reduced the catch of turtles, especially in the NPF after the introduction of Turtle Excluder Devices (Brewer et al., 2006). Turtles are significant to the Indigenous fishery in the GoC, with turtles and eggs harvested for food. Although data are not available for the GoC, the Queensland component of the 2000/01 NRIFS estimated that 3851 Marine Turtles and 3976 eggs were harvested.

Risk score justification

Marine Turtles are widely distributed and use a range of habitats throughout the GoC and do not rely exclusively on estuaries, as a nursery, for feeding or reproduction (Figure 2.70). They may be impacted by reduced flows through reduced abundance of invertebrate prey, although this may be minor. However, a reduction in flow may benefit the green turtle by reducing high turbidity flood events associated with high flows that lead to seagrass mortality. Two species of Marine Turtles show significant population subdivision throughout northern Australia and the GoC, although they reproduce year round, indicating that floods are unlikely to be a reproductive cue. Therefore, reduced flows are unlikely to have a detectable effect on the populations of Marine Turtles.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

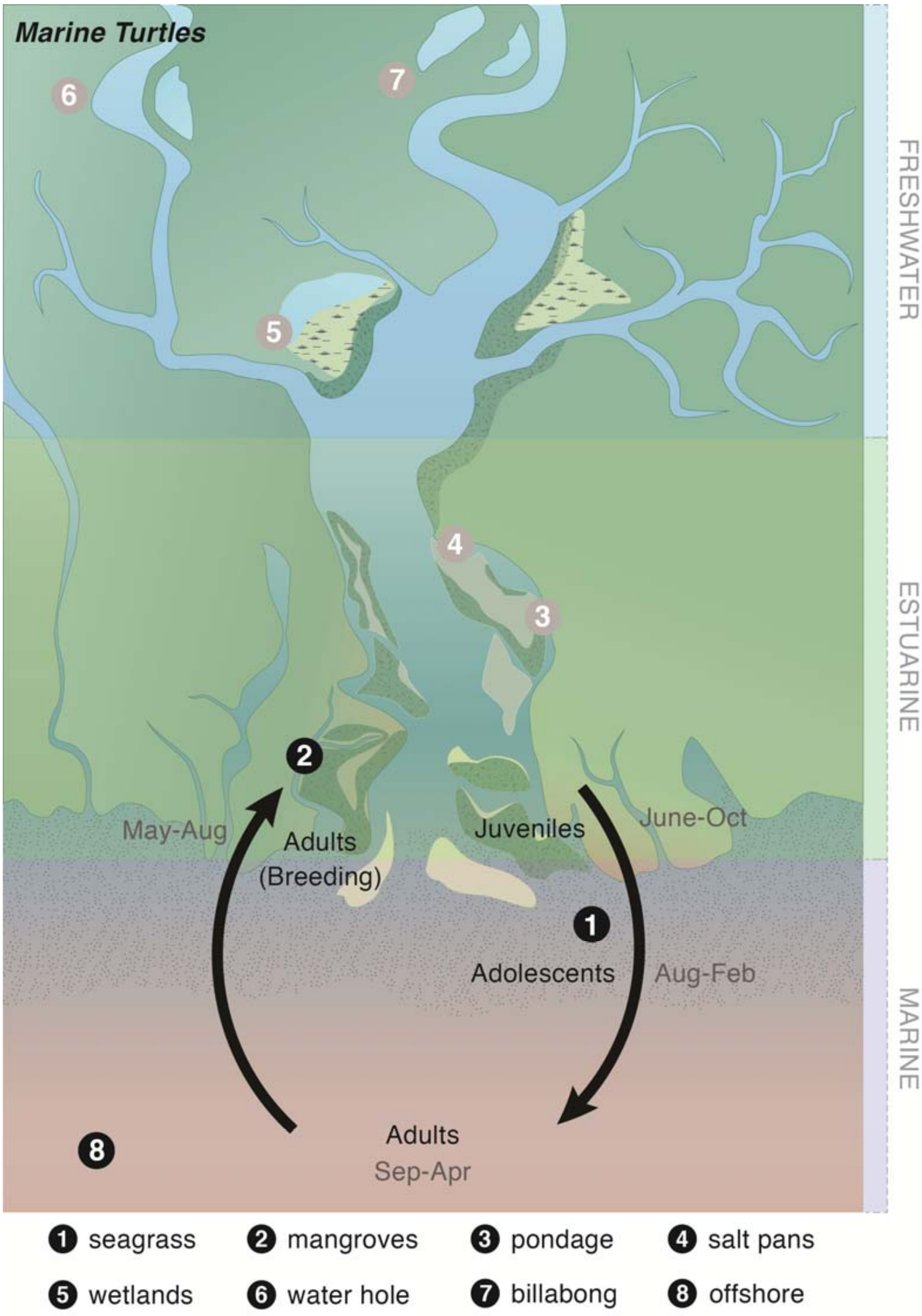

Figure 2.70. Conceptual model of the life history of Marine Turtles, illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.32 Dugong

The Dugong (*Dugong dugon*) is a large marine mammal found in shallow tropical waters throughout the Indo-Pacific (Marsh et al., 2002). The Dugong is under threat throughout most of its range, with fragmented populations, and is listed as 'Vulnerable' to extinction under the IUCN (Marsh et al., 2002; Marsh, 2008). Dugongs have high cultural, spiritual and social significance for Indigenous communities (EPA, 1999; Marsh et al., 2002).

Dugongs are long lived (~70 years) and reproduce every 3 to 5 years once maturity is reached (9 to 17years) (Marsh et al., 1984; Marsh and Dinesen, 2002; Pomeroy, 2011). Dugongs from northern Queensland breed in the second half of the year, and calve from August-December after a gestation period of some 12 months (Marsh et al., 1984). Shallow waters and estuaries have been reported as sites for calving, potentially to reduce predation risk from Sharks (Marsh et al., 1999; Whiting, 1999; Marsh et al., 2002).

A high proportion of Dugongs occur in shallow protected bays, wide shallow mangrove channels and in the lee of large inshore islands, although Dugong have been observed in offshore deeper waters presumably feeding on deepwater seagrass (Marsh et al., 1999). Major concentrations of Dugongs coincide with large extents of seagrass beds (Anderson, 1982; Bayliss and Freeland, 1989; Marsh et al., 2002). Dugongs have a highly specialised diet of seagrass and appear to prefer seagrass species with relatively high nutritional value, particularly *Halophila* and *Halodule* in northern Australia (Marsh et al., 1982; Marsh et al., 2002). Dugongs are estimated to consume 21 to 36 kg of seagrass per day (Marsh et al., 1999). Life history and reproduction in Dugongs are negatively affected by seagrass dieback (Preen and Marsh, 1995; Marsh and Kwan, 2008).

The tropical and subtropical coastal waters of Australia are believed to support most of the world's Dugongs (Marsh et al., 1999; Marsh et al., 2007). The total Dugong population along the Queensland coast of the Gulf was estimated to be 4266 in Dec 1997, around 5% of the estimated total number of Dugong in Australian waters (Marsh et al., 1999; Marsh et al., 2002). Aerial surveys in the 1970s revealed that the area around the in the Wellesley Islands was the most important Dugong habitat along the eastern coast of the Gulf (EPA, 1999). Queensland's third largest population of Dugongs occurs in this region, and it is likely the most important Dugong habitat in the Gulf (Marsh et al., 2002). However, more Dugong were found in the western Gulf than the eastern Gulf, presumed to reflect the greater areas of seagrass along the Northern Territory coastline (Marsh et al., 2002).

Surveys around northern Australia reveal considerable movement of Dugongs spanning hundreds of kilometres (Marsh et al., 2002; Marsh et al., 2007). Large-scale movements appear to be in response to a range of drivers such as seasonal changes in water temperature, movements among seagrass foraging habitats or catastrophic destruction of seagrass beds due to, for example, cyclones (Preen and Marsh, 1995; Preen et al., 1995; Sheppard et al., 2006).

Risk score justification

Dugongs are widely distributed and primarily utilise seagrass habitats (Figure 2.71) throughout northern Australia. Considering the biomass of seagrass is significantly less in the study region than other areas of the GoC (e.g. the southwestern and western GoC), the impact of altered flow is likely to be significantly less on Dugongs. Dugong may be indirectly impacted through reduced flow through changes to seagrass beds, although it may be more likely that Dugongs may benefit through expansion of seagrass extent due to reduced turbidity. Therefore, reduced flows are unlikely to have a detectable effect at the GoC population level of Dugong.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

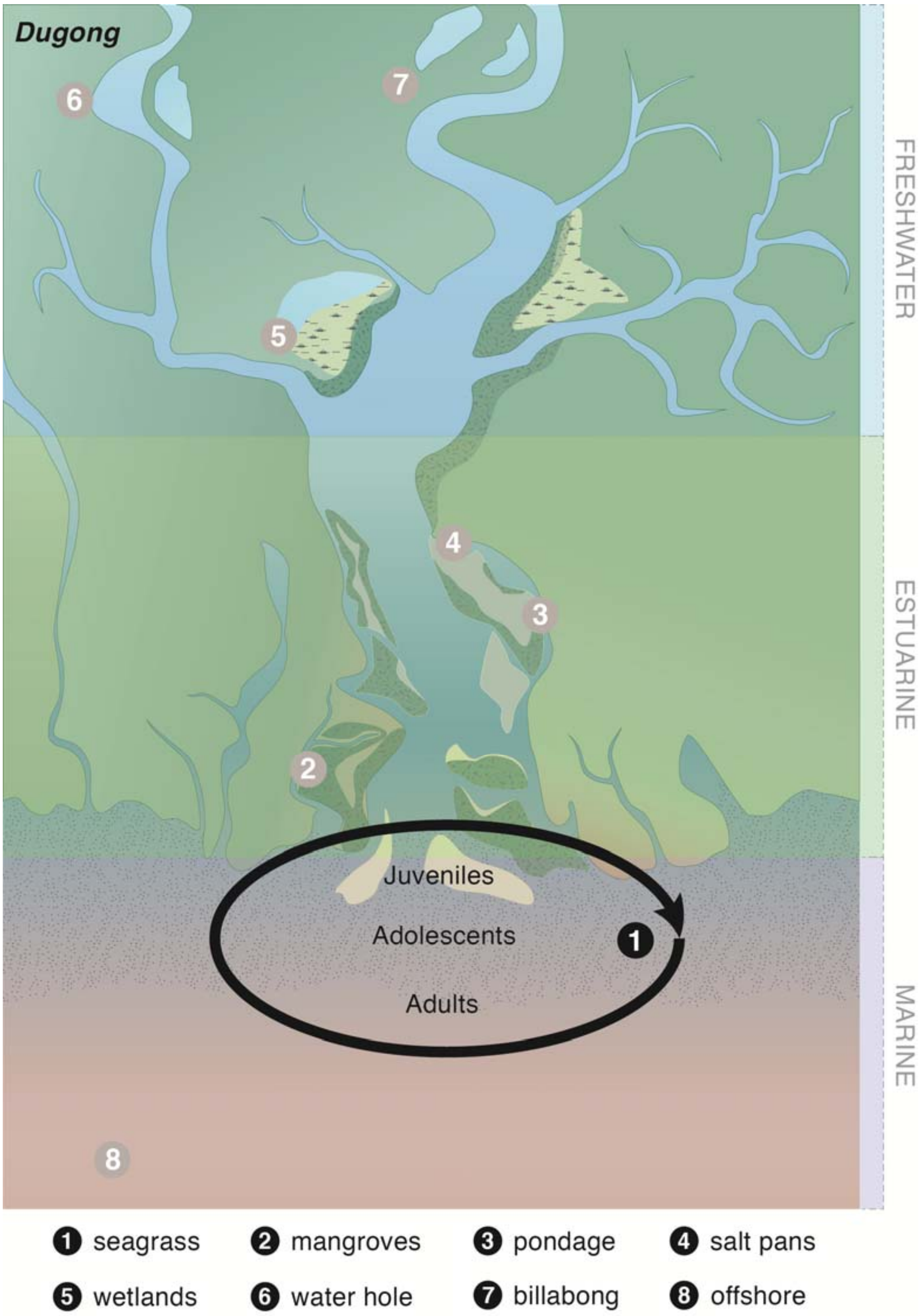

Figure 2.71. Conceptual model of the life history of Dugong, illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.33 Saltwater Crocodile

Two species of Crocodile occur in Australia, the Saltwater (estuarine) Crocodile (*Crocodylus porosus*) and freshwater Crocodile (*C. johnstoni*). This review focuses on the Saltwater Crocodile given their use of estuarine and coastal waters. Saltwater Crocodiles are found throughout coastal southern India and Sri Lanka, south-east Asia and as far south as northern Australia (Johnson, 1973; Webb et al., 2010). The species is listed as 'Vulnerable' in Queensland (EPA, 2007) although populations across northern Australia have shown some recovery following protection some 30 years ago from unregulated hunting (Read et al., 2004; Fukuda et al., 2007; Fukuda et al., 2011).

Saltwater Crocodiles are commonly found in mangrove swamps, coastal marshes, estuaries, rivers, and inland freshwater lakes and marshes (Bayliss et al., 1986; EPA, 2007; Brien et al., 2008; Webb et al., 2010). Saltwater Crocodiles also utilise coastal marine waters, and regularly disperse among rivers along coastlines and occupy offshore islands (Webb and Messel, 1978; Brien et al., 2008; Fukuda and Cuff, 2013). High densities of Saltwater Crocodiles are reported from freshwater swamps, and they predominately select freshwater habitats for nesting (Webb et al., 1977; Brien et al., 2008; Fukuda and Cuff, 2013). In the southern Gulf of Carpentaria, Saltwater Crocodiles can be found 100s of kilometres upstream (Read et al., 2004; EPA, 2007). They penetrate rivers as far as accessible (until major obstacles are reached) and thus are often found further inland during the wet season when flooding increases habitat extent and accessibility (Letnic and Connors, 2006). Females occupy a home range that extends in extent during the wet season as flooding opens new habitat while males move more extensively over larger territories, or show a 'nomadic' movement over large distances, particularly during the late dry/early wet season (Kay, 2004; Read et al., 2007; Brien et al., 2008; Campbell et al., 2013). Saltwater Crocodiles are not highly territorial, with overlapping home regions commonly observed (EPA, 2007). A higher proportion of smaller (juvenile) crocodiles tend to occur further inland in freshwater habitats than in estuarine environments where populations are dominated by larger individuals(Read et al., 2004; Letnic, 2008). During the late dry season, individuals > 1.2m long, aggregate above tidal river sections and move into flooded freshwater swamps during the wet season (EPA, 2007).

Analysis of Crocodile abundance surveyed in river catchments across northern Australia, including the Norman and Staaten rivers in the south-western GoC, with environmental factors indicated temperature, extent of favourable freshwater vegetation and rainfall seasonality (as a proxy for flooding and ecosystem productivity) were important (Fukuda et al., 2007). Favourable vegetated habitats in wetlands, swamps and floodplains showed particularly strong association with Crocodile density. The extent of favourable habitat for Saltwater Crocodiles in the eastern GoC, is higher north of the Embley River where extensive freshwater swamps with low risk of seasonal flooding are found (Magnusson et al., 1980). Further south, both the extent of favourable habitat and the abundance/biomass of Saltwater Crocodiles (sightings per kg per km river) decreases although crocodiles are still present (0.86 and 1.10 sightings per km for the Norman and Staaten rivers surveyed 150km above river mouths) (Magnusson et al., 1980; EPA, 2007; Fukuda et al., 2007). Eight habitat regions have been defined for Saltwater Crocodiles in Queensland, with the Flinders and Gilbert rivers falling in the Southern Gulf Plains and Northern Gulf Plains respectively (EPA, 2007). The only significant concentrations of Saltwater Crocodiles in the Southern Gulf Plains are found in the middle reaches of the Norman River (EPA, 2007). Surveys (1994-2000) of the eight Crocodile regions around the northern Queensland coast found that < 5% of hatchling crocodiles and < 14% of older crocodiles were observed in the Southern and Northern Gulf plain regions collectively (EPA, 2007).

Reproduction commences during the wet season (Webb et al., 1977; Webb et al., 1983; EPA, 2007). Courtship and mating are probably triggered by rainfall and the relatively cooler temperatures associated with the end of the dry season (Webb et al., 1977; Isberg et al., 2005). Breeding and recruitment occur principally in rivers with significant freshwater input or freshwater swamps (Webb et al., 1977; Fukuda and Cuff, 2013). Eggs (~50) are laid in nests composed of mounds of vegetation above the flood level either adjacent to rivers or in freshwater swamps (Webb et al., 1983; Thorbjarnarson, 1996). There are likely to be multiple factors influencing the placing of nests including the availability of deep freshwater as refuges for the females, that suitable nest sites are more available in freshwater habitats than saline habitats and suitability of ground vegetation for constructing nests (Webb et al., 1977; Magnusson, 1980; EPA, 2007; Fukuda and Cuff, 2013). Nests are rarely sited 100m or more from permanent deep water (Webb et al., 1983). The presence of freshwater appears to be important for Saltwater Crocodile reproduction. However, as nesting occurs during the wet season, egg mortality due to flooding can be high (Webb et al., 1977; Magnusson, 1982; Webb et al., 1983). Females may watch the eggs during the incubation period (70 to 90 days) and have been observed to assist hatchlings to reach water (Webb et al., 1977; Magnusson, 1979).

Saltwater Crocodiles feed on animals along the water's edge, birds from the water surface and fish and crustaceans (Johnson, 1973; Webb and Messel, 1978). Growth rates of juvenile crocodiles are fastest during the wet season when prey is abundant and flooded habitats are extensive (Webb and Messel, 1978; Magnusson and Taylor, 1981).

Risk score justification

Saltwater Crocodiles are widely distributed and use a range of habitats throughout the GoC, but primarily use estuaries, as a nursery, for feeding and reproduction (Figure 2.72). A reduction in wet season flow may reduce the extent of wet season habitat and food availability for local Crocodile populations and reduce habitat connectivity, particularly to suitable nesting sites for male Crocodiles during breeding season. However, a reduction in peak flows may benefit Crocodiles through a reduced risk of egg mortality through flooding. Therefore, reduced flows are unlikely to have a detectable effect at the GoC population level of Saltwater Crocodiles.

Risk scores: Consequence **1**; Likelihood **3**. **Overall risk rating**: LOW

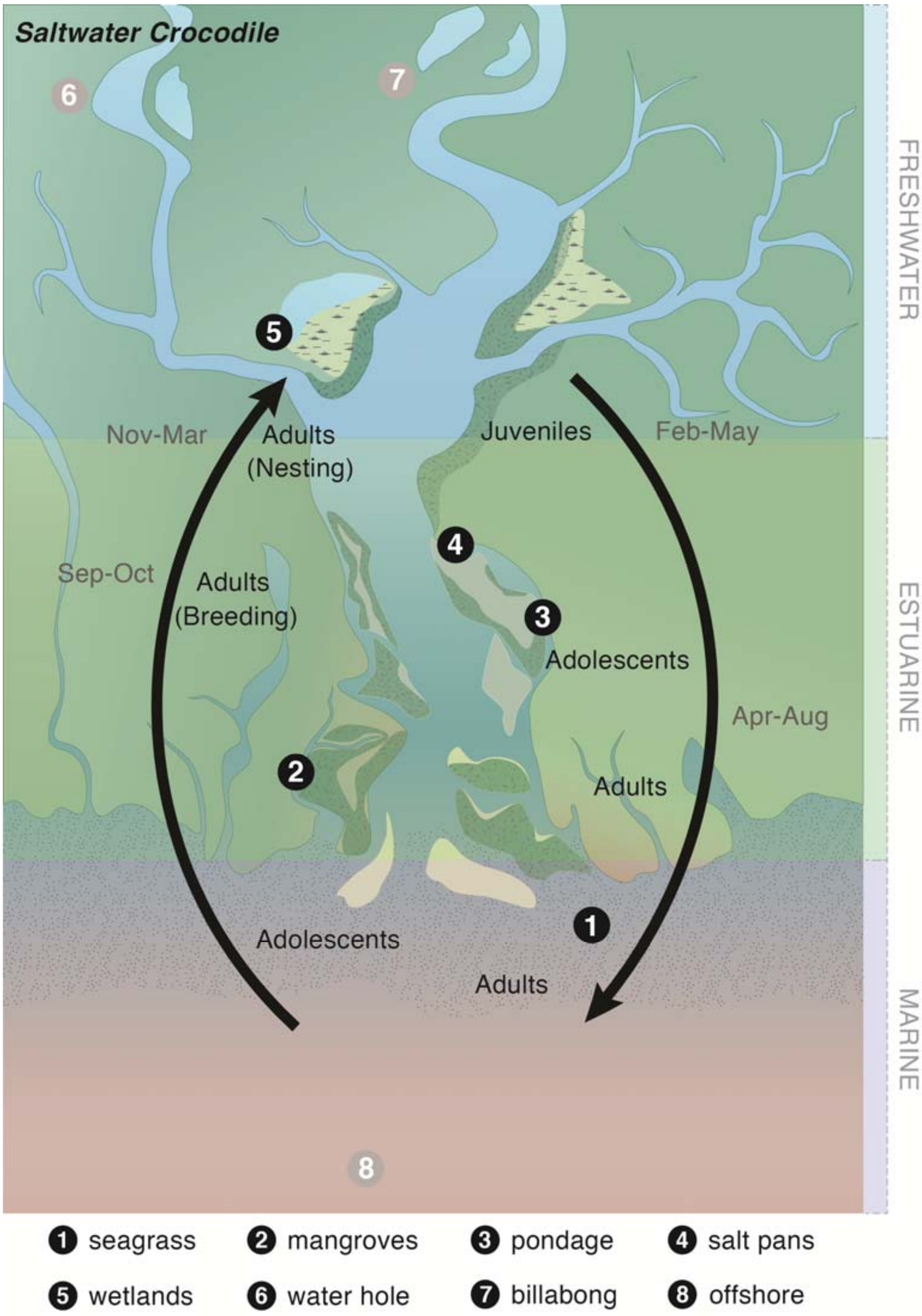

Figure 2.72. Conceptual model of the life history of Saltwater crocodile, illustrating its use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.34 Migratory Shorebirds

There are 34 species of migratory shorebird that spend their non-breeding season in the study area in the south-eastern GoC. They are classified into two families based on differences in morphology and ecology: Charadriidae (plovers, dotterels, lapwings) and Scolopacidae (curlews, godwits, sandpipers and knots). Species in both families migrate between their breeding grounds in the Arctic and countries in the southern hemisphere. The birds breeding in China, Mongolia, Russia and Alaska migrate to Australia and nearby countries and from the East Asian–Australasian Flyway (Figure 2.73). Over half of the eight million Migratory shorebirds in this Flyway spend their non-breeding season in Australia and New Zealand [4 to 5 million] (Bamford et al., 2008).

The south-eastern GoC is the third most important staging and non-breeding area for Migratory shorebirds in Australia and the most important in eastern Australia (80 Mile Beach and Roebuck Bay WA are 1 and 2). The region is a gateway for most Migratory shorebirds coming and going from eastern and southern Australia and New Zealand (Bamford et al., 2008; Driscoll, 2014). In the south-eastern GoC, Migratory shorebirds rely on a range of coastal and sub-coastal wetland habitats for feeding and roosting during transit and the non-breeding season. Mixed species can form huge aggregations on sand/mudflats at river mouths, creek margins, claypans and saltmarshes. Each species prefers a different suite of habitats that in turn depend on river and monsoonal flooding to maintain their productivity. Within the south-eastern Gulf of Carpentaria, most (~ 80 %) Migratory shorebirds congregate in the rich coastal zone 50 km east and west of mouth of the Norman River (Driscoll, 2014).

There are limited data on the distribution and abundance of Migratory shorebirds in the south-eastern GoC due to the remoteness and difficult access to coastal habitats. Two comprehensive surveys have been made, first in 1997-1999 and a second in 2013 (Driscoll, 2001; 2014). Wetlands International (2012) has found an alarming declining trend in shorebird numbers in the EAA flyway. A comparison of the counts of shorebirds within the study area between 1997-99 and 2013 reveals variable population trends among species but an overall decline of twenty-one percent (21%) in combined counts of the eight species of international significance (≥ 1% of their Flyway population) (Table 2.15). The causes of the declines in Migratory shorebirds have been attributed to habitat loss and hunting along the Flyway (Amano et al., 2010; Murray and Fuller, 2012). Migratory shorebirds rely heavily on benthic invertebrates that occur in higher abundance in and on the intertidal mud/sand banks near mouths of rivers and creeks. Water extraction has played a critical role worldwide in the submergence and loss of river delta area (Syvitski et al., 2009). In the East Asian-Australasian Flyway, over half of important coastal feeding habitats used during staging by over six million migrating shorebirds have been lost in Korea and China (Cho and Olsen, 2003; An et al., 2007; Murray et al., 2014). These effects are most dramatic in the Yellow Sea where the sediment transported to the deltas of the Yangtze and Yellow Rivers has reduced by up to 90% (Syvitski et al., 2009). This region also supports most Migratory shorebirds from Australia as they stage and build up energy reserves before flying to their breeding grounds (Rogers et al., 2010).

Due to their migratory behaviour, Migratory shorebirds have received heightened conservation protection. There are a number of international conventions to which Australia is a signatory that cover these species and their habitats. All species are covered under the migratory bird agreements with China, Japan and South Korea (JAMBA, CAMBA and ROKAMBA). For example, nine species of shorebird in the study area are present in internationally significant numbers (≥ 1% of a flyway's population) and thus meet one of the criteria to justify the region as a wetland of international importance (Ramsar, 2008) (Table 2.15). The EPBC Act also lists each migratory shorebird species as 'matters of national importance'. The Act requires any human activity that can potentially adversely affect their populations to minimize or mitigate the impact. Other international bodies such as the IUCN also undertake assessments of the population status and trend for all vertebrates, including Migratory shorebirds. These assessments apply objective criteria to assess the population trends for each species. They show that almost all Migratory shorebirds with large populations in the study area have been identified by their experts as declining and two (Great knot and Eastern curlew) are 'Vulnerable' (Table 2.15).

Risk score justification:
Reduced or regulated flow would impede sediment delivery to river mouths and therefore reduce the replenishment and productivity of feeding and roosting grounds on coastal mud/sandflats sub-coastal claypans (Figure 2.74). The proposed flow extraction scenarios would also reduce inundation of freshwater wetlands in the lower and middle Flinders and Gilbert rivers. The area and frequency of inundation of these wetlands would be especially affected during median or below median flow years. Several migratory and non-migratory (avocets, stilts, dotterels) species of shorebird depend on these habitats during their staging in the region. During their stay, they accumulate fat reserves to make the long flights to their next staging location in eastern Asia. The risk is also increased due to the incremental loss of coastal wetlands elsewhere in Australia and in the birds' important staging sites in eastern Asia. The intertidal areas on the coast of the Flinders and Gilbert rivers catchments are also the major last staging region for Migratory shorebirds from eastern Australia. Thus, the loss or degradation of feeding habitat in the study area will decrease the population viability during migration and subsequent breeding.

Risk scores: Consequence **3**; Likelihood **4**. **Overall risk rating**: HIGH

Table 2.15. The species of Migratory Shorebird in internationally-significant numbers counted in the 2013 survey of the study area, their estimated flyway population, population trend, IUCN status and the changes in absolute and percentage counts between the 1997–99 and 2013 surveys.

Common name	1997-99[1]	2013[2]	WPE4[3] x 10^3	WPE5[4] x 10^3	1997-99 F'way %[5]	2013 F'way %[6]	WI F'way Trend[7]	IUCN Status[8]	Δ 1997-99 to 2013[9]	Δ %[10]
Great knot	30,514	21,466	380	290	8.0	7.4	Declining	Vulnerable	–9,048	–42
Black-tailed godwit	9,389	9,200	160	140	5.9	6.6	Declining	Least concern	–189	–2
Red knot	9,976	5,661	220	110	4.5	5.1	Declining	Least concern	–4,315	–76
Red-necked stint	1,467	3,302	315	320	0.5	1.0	Unknown	Least concern	1,835	56
Greater sand-plover	822	2,595	100	79	0.8	3.3	Declining	Least concern	1,773	68
Broad-billed sandpiper	117	472	50	25	0.2	1.9	Unknown	Least concern	354	75
Eastern curlew	587	441	38	32	1.5	1.4	Declining	Vulnerable	–145	–33
Lesser sand plover	704	241	30	39	2.3	0.6	Declining	Least concern	–463	–192
Total	53576	43378								
Percent decline		21%								

[1.] 1997-99 mean abundance (Driscoll 2001) in survey areas E&F; [2.] 2013 mean abundance (Driscoll 2014) in survey areas E&F; [3] Wetlands International Waterbird EAA Flyway population estimates (WPS4) used to estimate flyway abundance in 1997-99; [4.] Wetlands International Waterbird EAA Flyway population estimates (WPS5) used to estimate flyway abundance in 2013; [5.] Percentage of species in EAA flyway found in 1997-99 surveys; [6.] Percentage of species in EAA flyway found in 2013 surveys; [7.] Wetlands International (2012) EAA Flyway population trends; [8.] International Union for Conservation of Nature (IUCN) conservation status; [9.] Change in mean abundance between1997-99 and 2013 surveys; [10.] Percent change in species abundance between 1997-99 and 2013 surveys.

Figure 2.73. The East Asian – Australasian Flyway showing the sites with internationally-significant counts of at least one species of Migratory Shorebird (Bamford et al. 2008).

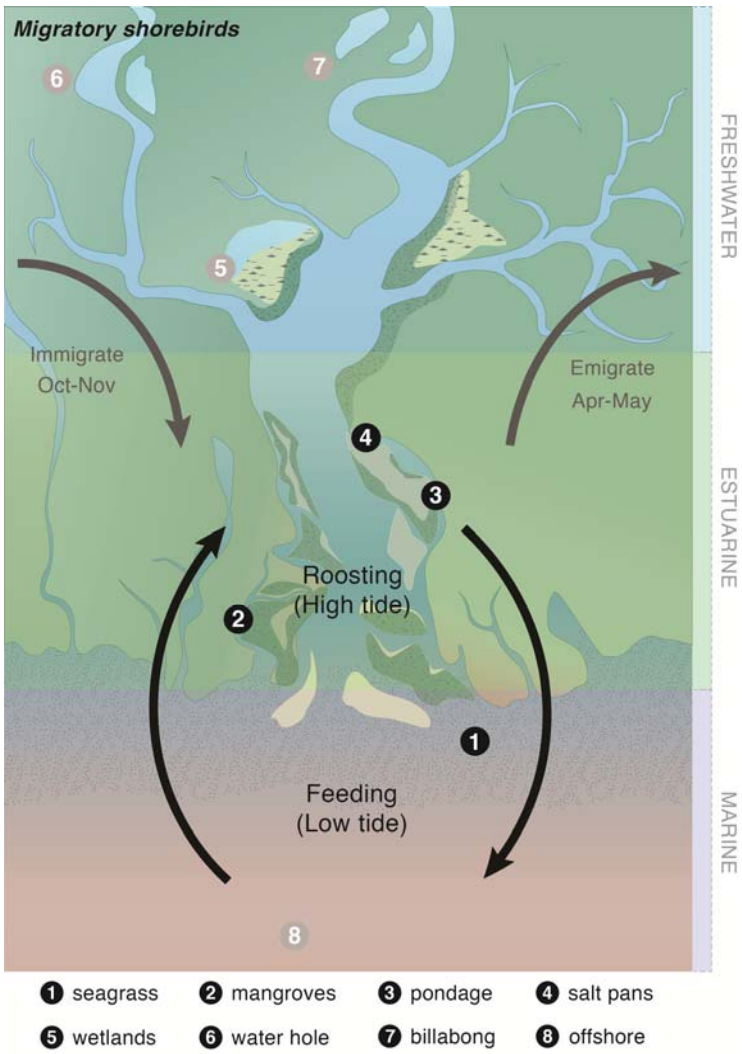

Figure 2.74. Conceptual model of the life history of Migratory Shorebirds, illustrating their use of marine, estuarine and freshwater habitats in the Gulf of Carpentaria. Numbers indicate habitat type shown in the key. Grey habitat symbols indicate habitat not used.

8.35 Mangroves

Mangrove communities are diverse assemblages of trees and shrubs that are found fringing most of the coastline of mainland Australia, with the most extensive and diverse communities found along the northern coastline (e.g. Wightman et al., 2004). For example, some 174,000 ha of mangroves (dominated by *Avicennia*) and saltpan occur along the coastline and estuaries between the Flinders and Gilbert rivers (Danaher and Stevens, 2000). A higher diversity of mangroves is found in wetter regions compared to drier coastlines, for example the eastern coast of Queensland versus the drier Gulf of Carpentaria (Bruinsma and Duncan, 2000), with volume of runoff from catchments being a contributing factor (Duke et al., 1998).

Mangroves grow in sediments that are generally low in oxygen and are highly saline and different taxa exhibit diverse morphological and physiological mechanisms to deal with such environmental stresses. Some species, such as *Rhizophora*, have aerial prop roots that branch down from the trunk and provide structural support in the mud soils. Others, such as *Avicennia,* have pneumatophores, aerial roots for gas exchange that grow vertically above the mud-soil surface from a shallow extensive network of anchoring roots. Mangroves act as a buffer between land and sea, filtering terrestrial discharge and decreasing sediment loading. They are valuable for their role in nutrient and carbon recycling and export of organic material to foodwebs in coastal waters. Importantly, their physical structure provides habitat and refuge for critical life history stages of fish and crustaceans taken by fisheries and Indigenous communities later in their life cycle (Jennerjahn and Ittekkot, 2002; Wightman et al., 2004).

Mangroves support numerous crustaceans, fish (many endemic to the GoC), and several other species such as penaeid prawns that form highly complex food webs (Danaher and Stevens, 2000; Laegdsgaard and Johnson, 2001; Vance et al., 2002; Manson et al., 2005). It was estimated that 75% of important commercial, and recreational fish and crustacean species in Moreton Bay use mangroves at some stage of their life cycle (Quinn, 1992) and a review of mangroves as a driver of fishery production substantiates this trend worldwide (Nagelkerken et al., 2000). Mangroves in Queensland waters are protected under the *Fisheries Act 1994*, with additional protection for some mangrove communities as Fish Habitat Areas (reserves) with regard to the sustainability of dependent fisheries (Bruinsma and Duncan, 2000; Danaher and Stevens, 2000).

Mangroves require freshwater input and many mangroves live close to their salinity tolerance levels, so any change in flow regimes will affect mangroves in some manner. Hydrology of mangroves is complex; tidal inundation, rainfall, groundwater seepage and evaporation all influence soil salinity and have a profound effect on mangrove growth and distribution. Freshwater flow into mangroves reduces salinity, increases the water content of soils and delivers sediments and nutrients and creates conditions that are favourable for plant production (Smith and Duke, 1987; Ball, 1998; Lovelock et al., 2012). However, the location of mangrove species on a coastline can be generally described by two factors: freshwater input and intertidal position (Duke, 2013). Location on the open coast or in an estuary will determine fresh water input, whereas position on the shore will determine the length of time the area is inundated by saltwater. A ‘sequence’ of mangrove species can generally be observed going upriver from the mouth of estuaries or inland from coasts, reflecting the ecophysiological response of the plants along salinity, tidal and other environmental gradients and disturbance regimes (Danaher and Stevens, 2000). Although considerable variability is observed in the patterns of occupation and number of mangrove species among estuaries across northern Australia (Bunt, 1996).

Extraction of water from rivers and subsequent changes to flow regimes may negatively impact the productivity and extent of mangroves (Roderstein et al., 2014). Minor reductions in flow regimes have led to massive mortality of mangroves (Blasco et al., 1996).

High flow levels and input of nutrients from land during the wet season can induce rapid growth in mangroves (Kathiresan et al., 1996). Alteration of flow regimes will change the delivery of sediments from land runoff into estuaries, and therefore the availability of mangrove habitat and mangrove growth (Duke and Wolanski, 2001; Lovelock et al., 2007). Sedimentation within wetlands changes the surface height of wetland soils relative to sea level as well as extending habitat for mangrove colonisation. Mangrove forests can grow rapidly on newly deposited sediments and that recovery from storms and other disturbances can be rapid, although some species are more resilient than others (Lovelock et al., 2012). Extraction of water from rivers and subsequent changes to flow regimes will thus negatively impact the productivity and extent of mangroves (Roderstein et al., 2014). Minor reductions in flow regimes have led to massive mortality of mangroves (Blasco et al., 1996).

In addition, commercially important fish and crustacean species are strongly linked to the area of mangroves and salt marsh in Australian estuaries (Lee, 2004; Manson et al., 2005) and habitat connectivity (e.g. layout of mangrove patches and length of creek systems) among habitats (Pittman et al., 2004b; Loneragan et al., 2005; Skilleter et al., 2005; Meynecke et al., 2008). Export of organic matter from mangroves may also be important in sustaining coastal food webs (Jennerjahn and Ittekkot, 2002; Melville and Connolly, 2005). Thus, modification of mangrove extent and connectivity of patches, or mangrove loss, due to water extraction or modification of flow characteristics, will impact on the economic value of dependent fisheries and associated biodiversity (Lee, 2004; Loneragan et al., 2005; Manson et al., 2005).

Risk score justification:
A reduction in the volume of wet season flow is likely to reduce the productivity (growth) and composition of mangroves and their extent and connectivity, particularly in upper reaches of estuaries and high intertidal. However, it is likely that these effects will not affect the broader population of mangroves throughout the GoC, as effects are mainly confined to the Flinders and Gilbert rivers, with some minor effects on the adjacent coastal zone. Using the risk score criteria for habitats in Table 2.9, the overall risk at the population level is most likely minor.

Risk scores: Consequence **1**; Likelihood **4**. **Overall risk rating**: LOW

8.36 Seagrasses

Seagrasses are flowering plants that occur in shallow coastal waters and estuaries, generally on soft sediments, and can form extensive stands (meadows) (Butler and Jernakoff, 1999; Carruthers et al., 2002). A high diversity and extent of tropical seagrasses are found across northern Australia (Butler and Jernakoff, 1999; Carruthers et al., 2002). In conjunction with their contribution to littoral primary production, seagrasses function as ecosystem engineers; they baffle water flow, stabilise sediments, regulate nutrients, are grazed by turtles and Dugongs and support fishery species largely by providing nursery habitat (Blaber et al., 1992; Blaber et al., 1995; Connolly et al., 1999). Seagrasses along the eastern coast of the GoC are sparse with few areas of extensive intertidal seagrass, given the influence of large rivers and drainage basins compared to the western coast (Poiner et al., 1987; Carruthers et al., 2002; Roelofs et al., 2005). No seagrass beds were observed in tidal wetlands directly north of the Gilbert River but dense seagrass beds were observed at the mouth of the Norman River (Danaher and Stevens, 2000). In the GoC the offshore distribution of commercial Tiger Prawns reflects the inshore distribution of seagrass nursery habitat (Staples et al. 1985). The abundance of Tiger Prawns decreases from west to east in the south-eastern GoC adjacent to Mornington Island and the Gilbert River (Kenyon et al. 2011), reflecting less extensive seagrass communities in the south-east and east.

The critical factors controlling seagrass growth are light, temperature and nutrients (Butler and Jernakoff, 1999). The maximum depth at which seagrasses occur is limited by light attenuation in the water column, thus water clarity. This results in high turbidity and excessive freshwater runoff during the wet season, hence pulses of turbidity and reduced salinity, can lead to damage and dieoff of seagrasses (Butler and Jernakoff, 1999; Longstaff and Dennison, 1999; Carruthers et al., 2002; Carr et al., 2012). Floodflows may ultimately restrict their occurrence, however, flows also carry nutrients that support seagrass productivity. Analysis of patterns of change in intertidal seagrass *Halodule uninervis* extent and biomass at the mouth of the Norman River, showed that periods of lower biomass were correlated with high temperature and reduced flow periods (Rasheed and Unsworth, 2011).

The dynamics of tropical seagrasses are modified by extreme flood and cyclone events, driving temporal variability in patterns of seagrass abundance. Small-leaved tropical seagrasses in northern Australia, such as *Halophila* and *Halodule* species are generally very resilient to changes in the environment (Butler and Jernakoff, 1999). *Halophila ovalis* in the south-east Gulf can only tolerate light deprivation for relatively short periods (weeks) whereas *H. pinifolia* shows high tolerance to light reduction (Longstaff and Dennison, 1999). Tropical species generally lay down seed banks in the sediments so populations can recover quickly from widespread mortalities such as cyclonic disturbance (Butler and Jernakoff, 1999; Carruthers et al., 2002). Extensive loss of seagrass beds can lead to dramatic impacts on grazers. For example, a cyclones and two major flood events in 1992 led to massive dieoff of seagrass in Hervey Bay, Queensland (Preen et al., 1995), with subsequent mass mortalities of Dugongs and turtles, apparently due to starvation (Preen et al., 1995).

Numerous studies associate productivity of fishery species to seagrass status (Poiner et al., 1987; Pittman et al., 2004a) also see discussion in (Connolly et al., 1999). Declines in seagrass have led to changes in associated fisheries (Connolly et al., 1999). Seagrasses provide critical settlement and nursery habitats for the postlarval and juvenile stages of the life history of commercially important penaeid prawns, as well as fish species, in the Gulf (Blaber et al., 1992; Blaber et al., 1995) They also provide food and habitat for various fish species (Blaber et al., 1992; Brewer et al., 1995) and grazing for turtles and Dugongs.

Risk score justification:
Seagrasses, particularly within or near estuaries, are likely to be affected by reduced flow or alteration of land management in adjacent catchments. Increased sedimentation, nutrification and turbidity associated with stream channel and landscape erosion due to diverted flows or irrigation are likely to reduce seagrass productivity and extent. However, if sedimentation is managed adequately, reduced flows are likely to decrease turbidity and may lead to an expansion of seagrass beds. Using the risk score criteria for habitats in

Table 2.9, the overall risk as the population level is most likely minor, given that impacts will most likely only be measurable locally and not at a broader GoC scale.

Risk scores: Consequence **1**; Likelihood **4**. **Overall risk rating**: LOW

8.37 Floodplains

The topography of the southern GoC is very flat. Therefore, in years with significant rainfall in the wet season, extensive flooding results across the catchments that may persist for months. Flooding stimulates growth of benthic algae on low-lying areas, which typically lie dormant in the soil until wet for a few days. This is the base of the food sources for fish and other species. Studies of rivers in the GoC have shown that inundated floodplains (Figure 2.75) are the main diet source for barramundi, including those caught in coastal and estuarine areas (Jardine et al., 2012). In contrast, some other species (e.g. Bony Bream) do little feeding on floodplains but instead use the floodplain for reproduction. Therefore floodplains contribute to fish production in a variety of ways (Pusey et al., 2011b). Relative to known species in a river system, species richness and the density of individual species on floodplains can be very high (Pusey et al., 2011b). It is less clear which areas of floodplain are most critical to fish feeding and breeding, and what are the minimum periods of flooding needed for maintaining sustainable fish stocks.

Flood duration is also important. A study of three river systems in the wet-dry tropics of northern Australia (Mitchell River, GoC), Fitzroy River (WA), Daly River (NT)) showed that flood period affects consumer resource use (Jardine et al., 2012). Rivers with a longer flood period, such as those in the GoC, have fish which are less coupled to resources within the river. This is because floodplains provide much of the food needed by fish, consistent with the studies of Jardine et al. (2012). It may also explain the high levels of biodiversity in the southern Gulf rivers even though local primary productivity in rivers is relatively low.

Floodplains also contribute nutrients to rivers as water levels drop and floodplains contract. This helps to support fish which may have migrated up from estuaries in the dry season, as shown by a recent study in the Flinders River by Faggotter et al. (2013) and in adjacent rivers by Hunt et al. (2012).

Risk score justification:
Floodplains are likely to be affected by reduced flows due to a decrease in the area and period affected by flood flows that stimulate growth of primary producers and that form the base of the food web. Using the risk score criteria for habitats in Table 2.9, the overall risk as the ‘population’ level is most likely moderate, given that impacts will be localised, is significant, and will most likely result in the habitat not being able to recover in the longer term.

Risk scores: Consequence **2**; Likelihood **4**. **Overall risk rating**: MODERATE

Figure 2.75. Inundated floodplain on the Flinders River system. Photo: M. Burford

8.38 Saltflats

Coastal saltflats or claypans are shallow coastal basins which may be only infrequently tidally inundated (Figure 2.76). Tropical northern Australia has extensive areas of these saltflats which remain relatively pristine. In particular, the southern GoC contains thousands of square kilometres of low-lying, saltflats which are mostly vegetation free, and are coated in a thick salt crust for most of the year. During large rainfall events when overbank flow occurs, or during sustained local rainfall, they may be flooded for extensive periods.

Studies within the Fisheries Research and Development Corporation project 2007/003 examined the effect of flooding on primary productivity for these saltflats (Burford et al., 2010). They found that wetting of saltflats results in the release of high concentrations of nutrients, and after seven to ten days when benthic algae become productive they provide a food source for higher trophic levels. For example, Metapenaeid prawns, a key food source for barramundi and other fish species, were found to access the saltflats in high numbers after flooding (Kenyon et al., 2012). The implication of this is that flooding durations less than seven to ten days do not result in sufficient benthic production for prawn and fish feeding.

Risk score justification:
Saltflats are likely to be affected by reduced flows due to a decrease in the area and period affected by flood flows that stimulate growth of primary producers and that form the base of the food web. Using the risk score criteria for habitats in Table 2.9, the overall risk as the 'population' level is most likely moderate, given that impacts will be localised, is significant, and will most likely result in the habitat not being able to recover in the longer term.

Risk scores: Consequence **2**; Likelihood **4**. **Overall risk rating**: MODERATE

Figure 2.76. Saltflat inundated. Photo: M. Duggan

9 Recommendations

Following the qualitative risk assessment of species potentially impacted by reduced river flows in the Flinders and Gilbert rivers, a number of recommendations are made to improve our understanding assessment and forecasting of the effects of these altered flow regimes.

A qualitative risk assessment approach was required in the current project due to a general paucity of reliable biological and ecological information on species in these estuarine systems. It is recommended that detailed biological and trophic studies and improved fishery reporting be undertaken that will allow more reliable quantitative population and ecosystem assessments.

Specifically, these studies would include:

- age and growth to determine length-at-age relationships,
- reproductive biology to determine timing of spawning, fecundity, and age at maturity,
- trophic studies to determine the trophic relationships between species and the variation of these relationships over time, particularly in relation to flow,
- population subdivision investigations of key species using genetics and biological markers (e.g. parasites) to determine the scale at which biological parameters need to be understood.
- tagging studies using conventional and electronic tags to determine the large scale movements of animals between estuaries and other habitats, and small scale movements within estuaries to better understand utilisation of habitats that are affected by connectivity due to reduced flows (e.g. water holes and billabongs).
- improved species-specific reporting in commercial fishery logbooks, especially potentially vulnerable Shark species (e.g. 'hammerheads' and 'whaler' Sharks) that is required if fishery CPUE is to be used as an index of abundance for future population assessments, which is common practice in fishery stock assessment.

By undertaking the further work detailed above, further quantitative assessment of the potential ecological impacts of altered river flows may be undertaken using more formal stock assessment methods. This would allow the impact on populations and associated fishery, social and cultural values to be quantified. Although primarily useful for species of economic importance, stock assessments do not allow exploration of the whole-of-ecosystem impacts of reduced flows. Ecosystem models, such as the GoC ecosystem model used to identify keystone species in this project, are a powerful way to examine the broader ecological impacts of environmental perturbations. These models are becoming increasingly flexible, with the capability of integrating hydrological models to drive ecosystem changes through specified flow regimes. These models can also be spatially explicit, allowing different habitats and regions of the modelled region to be examined in isolation, which would be particularly useful in examining the potential ecological effects of introducing a range of flow regimes in the Flinders or Gilbert rivers individually, or in unison.

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Part 3 Quantitative Risk Assessment for Banana Prawns and Barramundi

Peter Bayliss, Julie Robins, Rik Buckworth, Cathy Dichmont, Roy Deng and Rob Kenyon

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Summary

1. Two iconic fisheries species in the Gulf of Carpentaria, the White Banana Prawn (*Penaeus merguiensis*) and the Barramundi (*Lates calcarifer*), were chosen *a priori* for quantitative risk analysis given their socio-economic importance, their well established strong dependence on freshwater flows to maintain populations, and the ready availability of catch and effort data for the Northern Prawn Fishery (NPF) and the Queensland Gulf Inshore Fin Fish Fishery (QGF).
2. The socio-economic assessment end point for both species is catch (t) per season, as this is a strong indicator of the performance of the fisheries and potential loss of production and income due to reduced freshwater flow. In contrast, the qualitative assessment endpoints used in Part 2 of this report focus on biological maintenance of the species assessed.
3. The FGARA development scenarios entail water allocations above current entitlements (scenario A) for the Flinders and Gilbert catchments and provide a range of estimates for reduced flow to estuarine and marine ecosystems, both as a percentage of the mean and median (Flinders: 1-20% of the mean wet year flow and 3-41% of the median; Gilbert: 1-20% and 1-28% of the mean and median, respectively).
4. To increase the range of water harvests for simulation purposes, we included additional scenarios that assumed the maximum in-stream storage capacity of each B.1 and B.2 scenario was used, and the maximum of the five new entitlements for irrigation using off-stream storage (scenario B.3). We assumed also that all scenarios will not be mutually exclusive and so can occur in any combination.
5. Regression models were developed to predict Banana Prawn catch (t) in fishing zones 7-9 of the Gulf NPF from effort (boat days) and the combined End of System (EOS) wet year flow (ML) of the Flinders and Gilbert rivers. Zone 8 is directly opposite both catchments and more vulnerable to reduced flows.
6. The models were used to predict the reduction in mean Banana Prawn catch between 1970-2013 in each zone and for each FGARA development scenario. Uncertainty analysis was used to incorporate model error and the intrinsic variability of flow and effort, providing a range of estimates rather than a mean value. For example, in Zone 8 for the B.2 scenarios combined across catchments, a 2 to 8% reduction in catch is predicted. Where maximum storage capacity is assumed used, a 3 to 13% reduction in catch is predicted.
7. Regression models were also developed to predict Barramundi catch (t) from effort (boat days) and wet year flow (ML) in the Flinders and Gilbert rivers using the same modelling approach. For the Gilbert B.2 scenario that had the greatest water harvest and used expected annual yields, 50% of simulations predicted a 3 to 12% (mean 10%) reduction in catch. Where maximum storage capacity is assumed used, a 4 to 19% (mean 16%) reduction in catch is predicted. For the Flinders B.1 and B.2 scenarios that used expected annual yields, 50% of simulations predicted a <2% reduction in catch with negligible range (and similarly for their maximum storage capacities). Where we assumed in Scenario B.3 that the maximum entitlement is used for off-stream irrigation/storage, 50% of simulations predicted a 2 to 4% (mean 3%) reduction in catch. The results for the Flinders should be treated with caution, however, as most (90%) observed catches are explained by fishing effort compared to flow, reflecting a slowly recovering fishery from decadal-scale droughts during the 1980s concomitant with heavy fishing.
8. The risk to Barramundi population recruitment from FGARA development scenarios as indexed by their Year Class Strength (YCS) was quantified also, given that DAFF data were available and it had previously been used in the Environmental Assessment of the Wet Tropics Water Resource Plan. Local relationships (equations) between Barramundi YCS and flow were developed for the Flinders and Gilbert rivers and then used to predict changes in YCS under pre-development (Natural), current (Scenario A) and proposed full-use water development scenarios (i.e. Flinders Scenario B.3 max and Gilbert Scenario B.2 max). Barramundi populations were considered at high risk of local population failure if modelled YCS was less than the median YCS for the modelled pre-development flow sequence for greater than 11 consecutive years. If the modelled YCS was less than the median YCS for 5 to 11 consecutive years then populations were considered at medium risk.

9. Results for YCS in both rivers show that no years were classified as high risk for Barramundi populations under pre-development and current water use scenarios. In contrast, under full-use water scenarios the high risk category increased from 0 to 3% in the Flinders River, and from 0 to 12% in the Gilbert River. Whilst Barramundi have evolved to deal with the flood-drought cycles of northern Australia, in a small number of years the proposed full-development water scenarios may artificially extend natural periods of low-flow to the estuary providing unsuitable conditions for recruitment and, therefore, population replenishment.
10. The catch-flow and YCS results are consistent in that they both demonstrate the relative effects of reduced flow are four times greater in the Gilbert than the Flinders. However, under full use water development scenarios the moderate YCS risk category increased from 10 to 24% in the Flinders River and from 5 to 26% in the Gilbert River. In contrast to the catch-flow model results, this suggests that the long-term sustainability of the Barramundi fishery could be at risk unless mitigation strategies are adopted (see Part 4).
11. Catch weight would not reflect the true socio-economic impact to both fisheries from sustained reductions in flow. We therefore strongly recommend that a detailed socio-economic assessment be undertaken to complement our first pass quantitative assessment of the potential risk of FGARA development scenarios for just two species in the Gulf fisheries. Inshore net fishers, for example, rely on more species than Barramundi for income so their potential economic loss would be greater.
12. We recommend also more detailed examination of the relationships between the extent, frequency and duration of floodplain flooding and fisheries production in both river systems, as their importance cannot be captured by total wet year flow volume. This should be complimented by further population level studies through YCS analysis of Barramundi and including King Threadfin, particularly for the Gilbert River that lacks such data. They dominate the catch in both rivers and, given their strong relationship with flow, would make ideal indicator species for monitoring potential impacts from sustained water extractions for agriculture as well as validating that the Thresholds of Concern used in the opportunistic YCS analysis summarised above appropriately represent the risk to Barramundi and associated fisheries. Additional monitoring of catch and movement of estuarine fish would also be beneficial in understanding lateral and longitudinal connectivity, within the Flinders and Gilbert catchments, and between adjacent catchments.
13. Other risk-based decision support tools need to be developed that incorporate the range of expert opinions normally associated with perceived risks, such as Bayesian Belief Networks, and to assess, evaluate and then communicate the effectiveness of proposed mitigation strategies.

11 Introduction

11.1 Background

Part 2 of this report provides a comprehensive qualitative risk assessment for those species in the Gulf of Carpentaria (GoC) fisheries identified as being at risk from freshwater extraction in the Flinders and Gilbert catchments for agriculture as outlined in the FGARA development scenarios (Part 1 Section 3; see Petheram et al. 2013a&b). Qualitative assessments also were undertaken in Part 2 for those species in the marine and coastal environment of the GoC that have known ecological, recreational or cultural values.

The qualitative assessments first identified species likely to be at risk from water harvest given their known life history (i.e. the consequences or effects). Conceptual models were then developed that captured the relationships between key stages in their life history and freshwater flow and, given the likelihood of exposure (i.e. up to 10% reduction in mean annual flow in both rivers, or up to 20% of median annual flow to account for the importance of low flow events; Petheram et al. 2013a,b). The qualitative risk assessment process (Part 1) identified a total of 46 taxonomic groups and four habitats that were considered by scientific experts and stakeholders on the project Reference group as being potentially at risk from reduced river flows in the Flinders and Gilbert rivers. Of this potentially vulnerable group, 15 (33%) were assessed as being at high risk from FGARA development proposals, 8 (17%) at moderate risk, 13 (28%) at low risk and 10 (22%) at negligible risk (see Part 2, Summary). The qualitative assessment also considered potential indirect risks to taxonomic groups of reduced flow arising from potential negative ecosystem effects on habitats using the EcoPath modelling framework (see Walters et al. 1997). Mangrove and Seagrass habitats were assessed as being at low risk from sustained reduced freshwater flows and, in contrast, coastal Floodplains and Saltflats were assessed as being at moderate risk.

A Best Practice risk assessment processes for an issue of this level of importance would be to follow up on the 'filtering' results of the qualitative assessments with more comprehensive quantitative assessments where appropriate data exist (see Bartolo et al. 2012; Bayliss et al. 2008, 2011), particularly for the 16 species/taxonomic groups identified as being at high risk. However, it was not possible to do this given the short five month timeline and broad scope of the project. Hence, two commercially important 'iconic' fisheries species, the White Banana Prawn (*Penaeus merguiensis*) and the Barramundi (*Lates calcarifer*), were chosen *a priori* for more detailed risk analysis given their socio-economic importance to the GoC region (Dichmont et al., 2008, Skirtum and Vieira 2012), their well established strong dependence on freshwater flows to maintain populations and therefore production (For Banana Prawns see Staples 1984; Staples and Vance 1986, 1987; Vance 1991; Vance et al. 2003; Venabales et al. 2011; and for Barramundi and other species see Loneragan and Bunn 1999; Robins et al. 2005, 2007a,b; Robins and Qifeng 2007; Halliday and Robins 2007; and Halliday 2012, 2008, 2011), and the ready availability of fisheries catch and effort data.

Part 3 of this report presents a quantitative risk assessment for these two important commercial species and is subject to the usual constraints and caveats when standard fisheries catch-effort data for population-level analysis are used (Harley et al 2001). Halliday et al. 2008 and 2012 advocate use of Year Class Strength (YCS) models to better ascertain relationships between commercial fish catch and river flow in Queensland fisheries, to overcome potential bias in catch-effort data where fish movements and flow may be confounded. The socio-economic assessment end point for both species is catch (t) per fishing season, and this is likely a strong indicator of fishery performance and associated potential loss of production and income due to reduced freshwater flow regimes (Dichmont et al. 2008, Pascoe et al. 2010).

There are four sections in Part 3. The first characterises the hydrology of the Flinders and Gilbert rivers as this is the source of freshwater into the GoC fisheries that sustain current production levels. In the second section we assess the potential impact of reduced flow from FGARA agricultural development scenarios for Banana Prawns, beginning with a characterisation of the NPF in the GoC followed by a quantitative risk assessment using statistical models that predict catch from fishing effort and flow. Similarly, the third section assesses commercial Barramundi catch in the Flinders and Gilbert rivers using the same quantitative approach. In addition, the risk to Barramundi population recruitment from FGARA development scenarios as indexed by their Year Class Strength (YCS) was quantified also, given that DAFF data were available and it had previously been used in the Environmental Assessment of the Wet Tropics Water Resource Plan. The fourth section discusses the results in relation to our overall assessment for the GoC. Part 4 Mitigation recommends future research and monitoring needs to support the Water Resource Plan for the GoC (Gulf WRP 2007).

11.1 The Flinders and Gilbert catchments

The key ecological driver in this assessment is the quantity and regularity of freshwater flowing into the GoC receiving waters adjacent to the Flinders and Gilbert catchments. We used the FGARA Source model outputs for Scenario A (historical plus current extractions/entitlements; FGARA 1890-2010) to characterise wet year (Oct-Sept) flow conditions in both rivers for the two time series used in the fisheries assessment (NPF 1970-2011; QGF 1989-2010).

The Flinders and Gilbert catchments comprise just two of 25 Australian Water Resources Council (AWRC) catchments that drain into the GoC (Figure 3.1). They are themselves separated by the Norman catchment. The surrounding catchments also contribute significant flows to the marine receiving waters of the GoC and these comprise mainly the Staaten and Mitchell to the north of the Gilbert, and the Leichhardt, Nicholson and other western GoC catchments including the Roper. The contribution to the marine fisheries in the GoC of these additional sources of flow is unknown but can be estimated using earlier Northern Australia Sustainable Yields (NASY) rainfall-runoff model outputs from 1930 to 2007 (Petheram et al. 2009a,b).

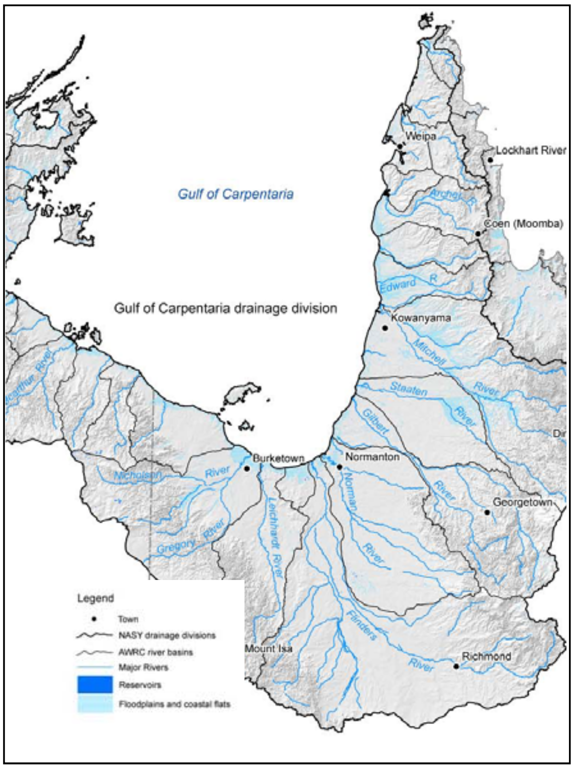
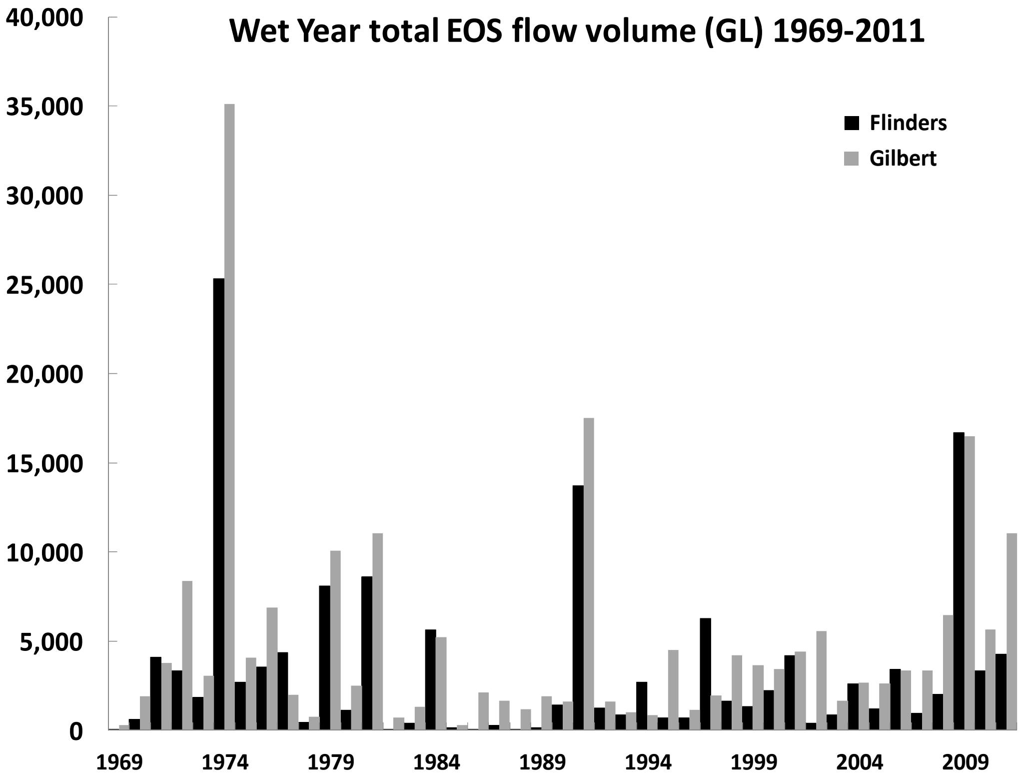


Figure 3.1 Catchments (AWRC river basins) in the Gulf of Carpentaria (modified from Fig. 3 in Petheram et al. 2009a).

Halliday et al. (2012) arbitrarily defined a wet year (or water year) in their report on flow impacts on estuarine finfish fisheries in the GoC as the months from October to September to encompass wet season rainfall and flow periods, and this definition is used here to allow comparison with their results. The wet season typically commence in November and finishes by April. The Gulf WRP and FGARA use July to June as the wet year, possibly because of drier rainfall conditions than elsewhere in the northern Tropics (J Robins pers. comm.) but, nevertheless, the wet season rainfall and stream flow conditions are essentially captured in the months October to September as in July to June.

(a)



(b)

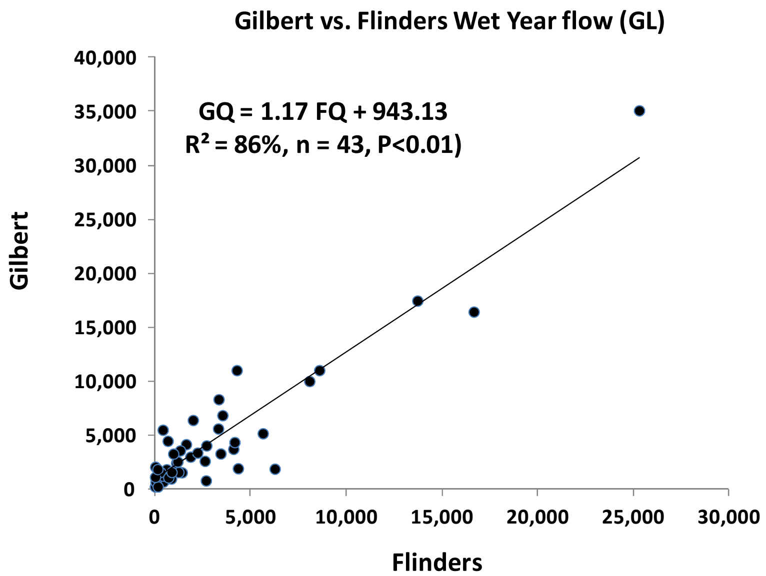


Figure 3.2 a&b (a) Trend in Source model "wet year" (Oct-Sept) EOS flow volume (GL) for the Flinders and Gilbert rivers, Gulf of Carpentaria, between 1969 and 2010 encompassing the period of the NPF. (b) Regression between EOS flow (GL) of the Flinders and Gilbert rivers.

The trends in the Source model EOS wet year flows (GL) for the Flinders and Gilbert rivers between 1969 and 2011 are illustrated in Figure 3.2a and encompass the period of the NPF showing very high peaks in the mid-1970s. This reflects a national trend in extreme rainfall and flooding events. The Gilbert and Flinders rivers generally show concordant patterns of wet year flow being adjacent catchments either side of the Norman catchment (Figure 3.2b, R^2=86%). In this time series low flow periods were generally more common than high-flow periods.

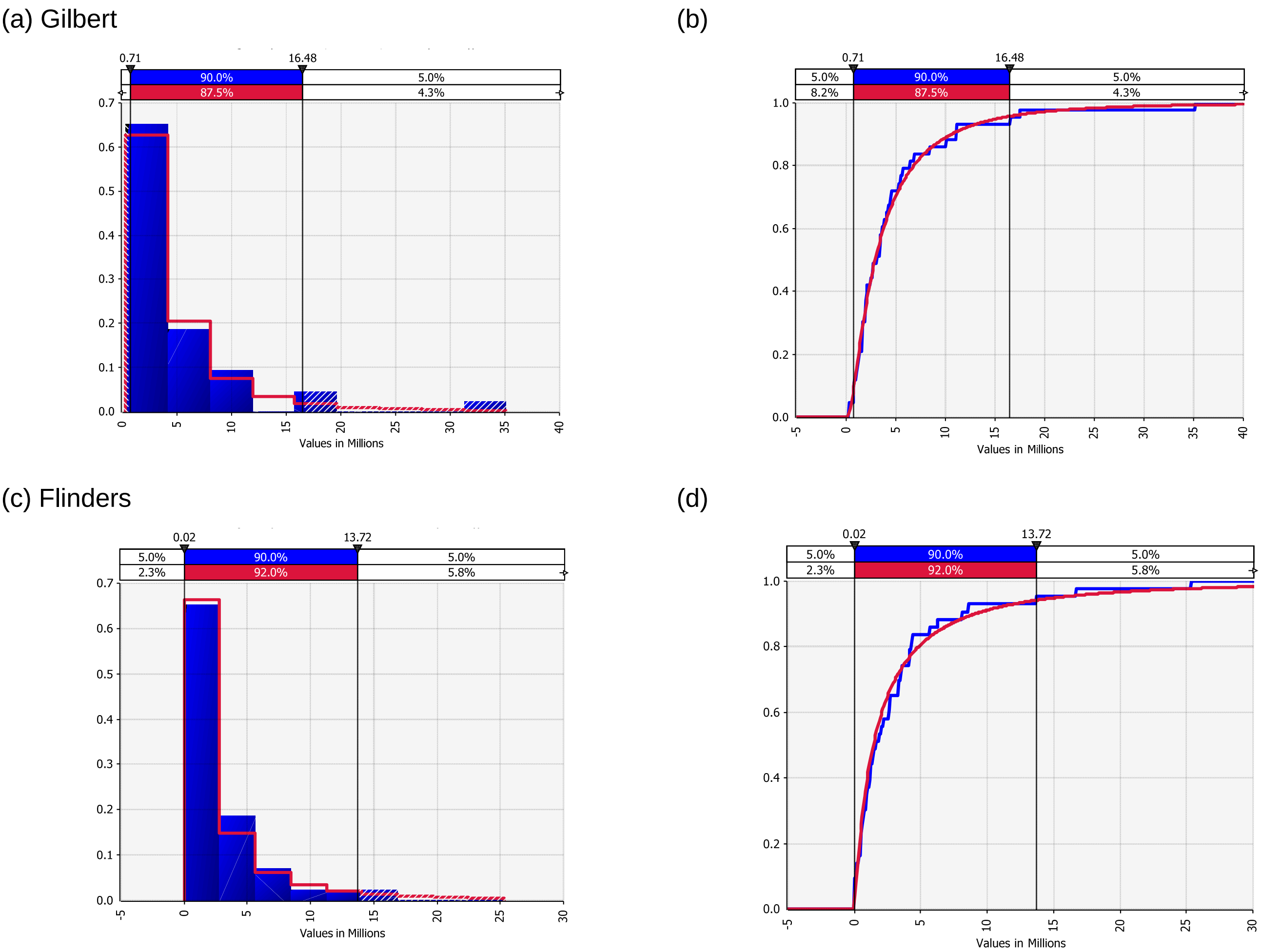


Figure 3.3 a-d Characterisation of End of System (EOS) wet year flow volume (ML) of Scenario A using "Best Fit" statistical functions (@Risk™ 2010). Blue = observed flow and Red = the relative frequency of lognormal flow.

The statistical properties of the wet year (Sept to Oct) EOS flow data derived from the Source model for the Flinders and Gilbert rivers, and for the different periods used in the GoC fisheries assessments, are summarised in Table 3.1. The flow data pertinent to FGARA and the fisheries are derived from different parts of the same time series, and thus have different statistical attributes. The FGARA time series models EOS wet year flow between October 1889 and September 2010 (n=121y), that for the NPF between October 1969 and September 2010 (n=41y), and for the QGF time series between October 1988 and September 2010 (n=22y). The data are non-normally distributed due to the high frequency of low flow periods and the low occurrence of very high rainfall-flow events, and may be characteristic of stream flow data in variable low rainfall environments such as the semi-arid southern GoC. The median value is about 30-50% of the mean value in all time series summarised in Table 3.1 (and Fig. 3.3 a & b), indicating that, depending on purpose, median flows may better define management objectives rather than the mean.

Flow data are intrinsically uncertain because they exhibit natural variability from floods to drought (Kennard et al. 2010a) and, which will be reflected in the frequency distributions of their class size intervals (Bayliss et al. 2011). The probability distribution, or probability density function (pdf), of a random variable is the statistical term for a frequency distribution constructed from an infinitely large set of values where the class size is infinitesimally small (Palisade 2010). Hence, the frequency distribution of all variables used in assessments over class size intervals is converted to continuous probability distributions that can be described by “Best Fit” equations chosen from a large range of candidate equations (Palisade 2010). These pdfs can be used to characterise a non-normally distributed variable more appropriately than its mean value assuming a normal distribution, hence they are used here to characterise flow. Best Fit pdfs were determined for the Flinders and Gilbert rivers EOS flow data for the time periods that encompassed the NPF and the QGF. Figure 3.3a-d shows the relative frequency and cumulative probability distribution functions for the Gilbert and Flinders rivers fitted to lognormal distributions, and which are used in all sensitivity and uncertainty analyses of risk models by transforming flow data to $\log_{10}$. Lognormal distributions appropriately characterise the distribution of low-flow data (see Brogan and Cunnane 2005).

Table 3.1 Summary of wet year (Sept to Oct) flow characteristics for the different periods of Source model EOS flow data (ML) used in the Gulf of Carpentaria fisheries assessments, as reflected in the statistical properties of their frequency distributions. The values at 10% are by definition values at low flow periods.

Variable	Gilbert			Flinders		
	FGARA	**NPF**	**QGF**	**FGARA**	**NPF**	**QGF**
	1890-2010	**1970-2011**	**1989-2010**	**1890-2010**	**1970-2011**	**1989-2010**
Years	**121**	**42**	**22**	**121**	**42**	**22**
Mean	3,685,417	4,840,141	4,216,136	2,669,635	3,289,535	2,988,270
Standard Deviation	4,238,694	6,153,499	4,349,634	3,783,967	4,878,024	4,146,032
Skewness	4	3	2	3	3	3
Kurtosis	28	17	9	15	13	9
Median	2,578,009	2,911,014	3,147,389	1,381,624	1,477,331	1,260,294
Minimum	62,581	245,850	760,096	0	2,430	79,825
Maximum	34,897,468	34,897,468	17,308,264	25,056,326	25,056,326	16,434,214
Count	121	42	22	121	42	22
10th percentile	226,206	957,556	1,039,185	1,191	110,130	603,929
90th percentile	7,784,775	10,785,965	6,284,667	7,123,144	7,881,619	6,112,740

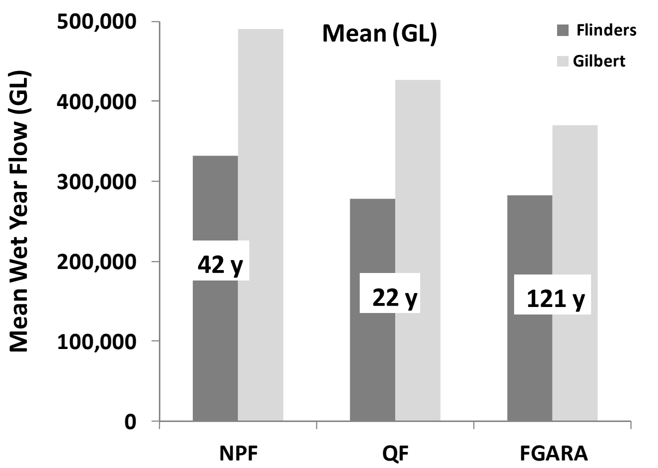
NB: a ‘wet year’ commences in the previous year, for example the 1890 wet year is from October 1889 to September 1890. FGARA = Source model period for Scenario A (historical plus current development/entitlements; Petheram 2013a,b). NPF = Northern Prawn Fishery, QGF = Queensland Gulf Fishery (or the Gulf of Carpentaria Inshore Finfish Fishery/GOCIFF plus the Mud Crab fishery). NPF data includes the 2011 wet year as the fishing season (April-June) is only two months short and in the dry season.

11.1.1 CLIMATIC DRIVERS – SEASONAL AND DECADAL TRENDS IN RIVER FLOW

Catchment hydrology is a major driver of coastal-marine ecosystems because of the connectivity between rivers and sea (Warfe et al. 2011). Hence, seasonal, inter-annual and decadal patterns of surface flow and associated ‘flood events’ are examined in relation to EOS flow over the FGARA 121y time series (Oct. 1889 to Sept. 2010) (Figure 3.4, Figure 3.5). River flow regimes in the GoC are highly seasonal with 99% of all flow volume (1889 to 2010) of the Gilbert River occurring during the wet season (October to April), and 0.7% in the Dry (May to October), 5.4% early wet (October to December), 89.2% mid wet (January to March) and 6.9% late wet (April and May). For the Flinders River 96% of all flow volume (1889 to 2010) occurred during the wet season, 4.0% in the dry season, 2.5% in the early wet, 86.2% in the mid wet and 7.1% in the late wet.

Distribution free cusum analysis (cumulative sum of the mean deviations; Manly and MacKenzie 2003) of EOS flow of the Flinders and Gilbert rivers combined exhibit an approximate 20-25y period nested under a 60-70y period (Figure 3.5a), a result also found by Bayliss et al. (2008) for several streams in the ‘Top End’ of the Northern Territory that have little or no water extraction. However, cusum analysis tends to enhance longer-term trends at the expense of shorter-term trends, hence Fourier analysis (Brockwell and Davis 1996) was used also to determine spectral signatures in the Source model flow data (n=121y, 1890-2010), and to ascertain average return periods more precisely. The periodogram of spectral density (smoothed values in Figure 3.5b) shows two large spikes at 4 and 10y, and a peak at 20-30y. Whilst the spikes at 4-10y most likely reflect a large component of white noise, they also correspond to periods of high El Niño–Southern Oscillation (ENSO) activity. A plot of smoothed periodogram values on period suggests a longer 60y period. The 'average' decadal (20-30y) trends in EOS flow for the Flinders and Gilbert rivers indicates that different time windows of flow data will encompass different flow characteristics that sum over the histories of above and below average rainfall years (Table 3.1). Hence, when assessing impacts of reduced flow using different and sometimes overlapping historical time series flow data, the frame of reference for the assessment conditions must be clearly stated.

(a)



(b)

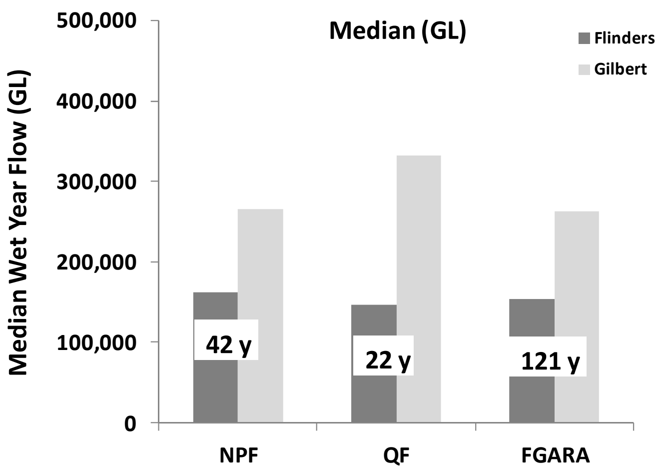
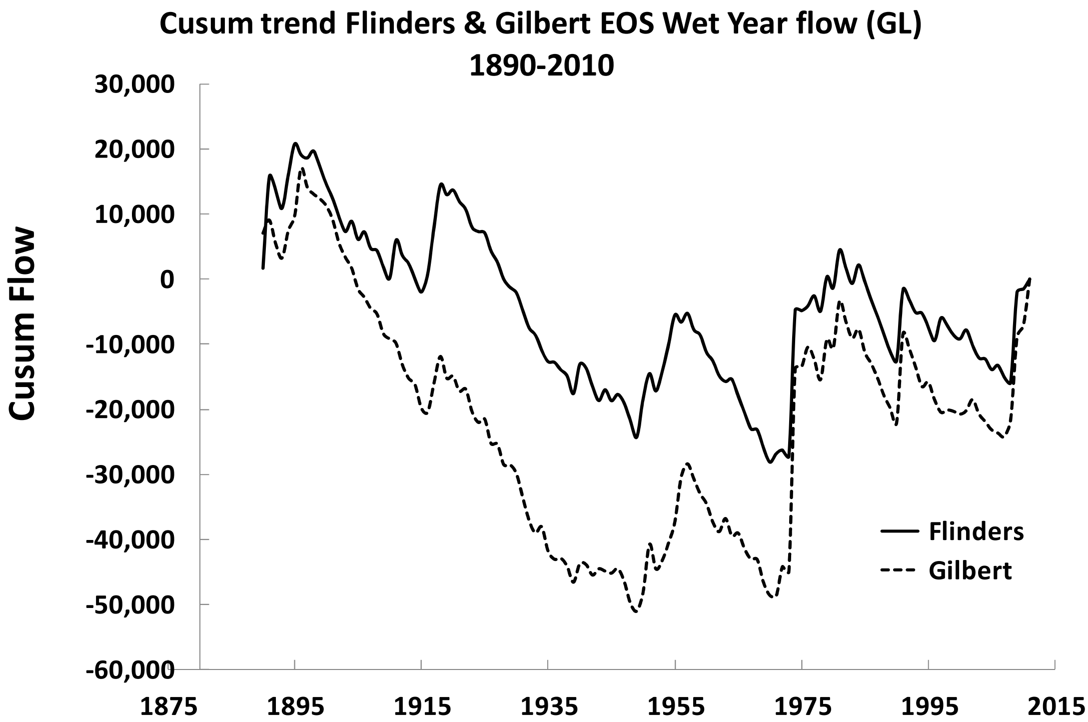


Figure 3.4 a&b (a) Mean wet year (October to September) EOS flow volume (ML) for the Flinders and Gilbert rivers as estimated from Source models (Petheram 2013a,b) for three times series used in assessments: FGARA – 1890-2010, n=121y; NPF – Northern Prawn Fishery 1969-2010, n=42; and QGF – Queensland Gulf Fishery (Inshore Fin Fish Fishery, 1988-2010, n=22y. In contrast the values for the (b) median were significantly lower (30-50% of mean value).

(a)



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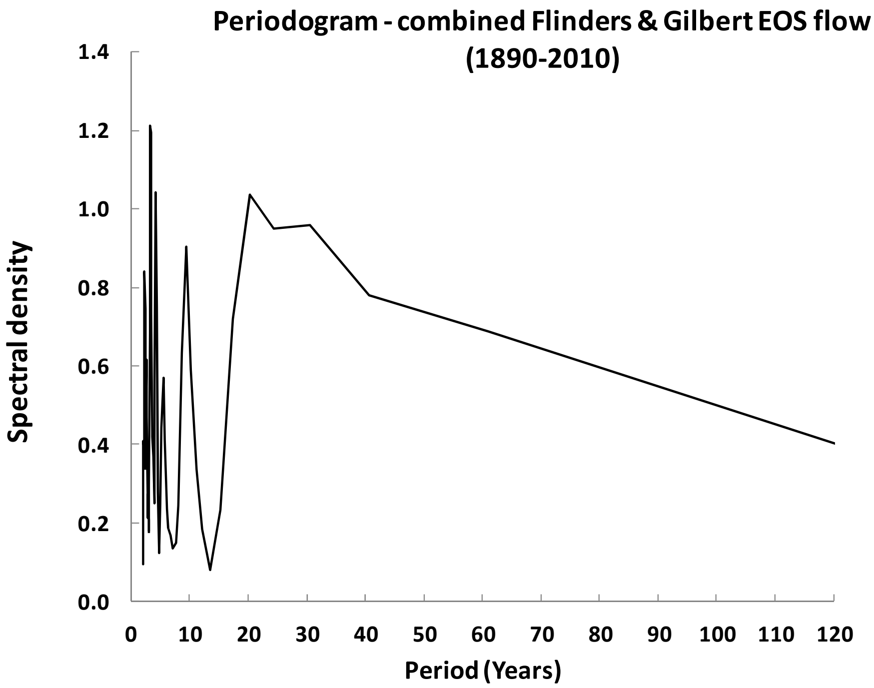


Figure 3.5 a&b. (a) Cusum trends (cumulative sum of the mean deviations of EOS wet year flow (ML) for the Flinders (solid line) and Gilbert (dashed line) rivers illustrating centennial and decadal trends. (b) Spectral analysis (periodogram) showing 20-30y periods in the Source model simulation data (natural flow) based on observed wet year rainfall (n=121y, 1890-2010).

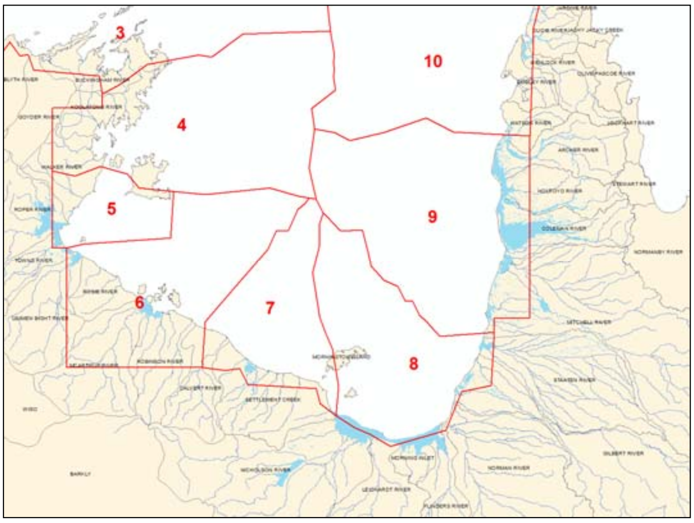
12 The Northern Prawn Fishery

12.1 Methods

12.1.1 CHARACTERISING CATCH AND EFFORT

The Northern Prawn Fishery (NPF) encompasses 10 fishing zones across northern Australia. It is characterised in Part 2 of this report (Section 7.1) and described elsewhere in detail (e.g. Dichmont et al. 2001).

(a)



(b)

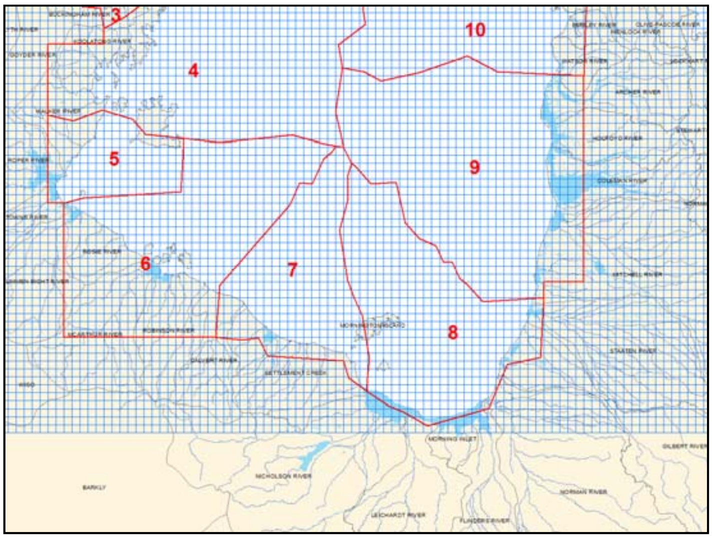
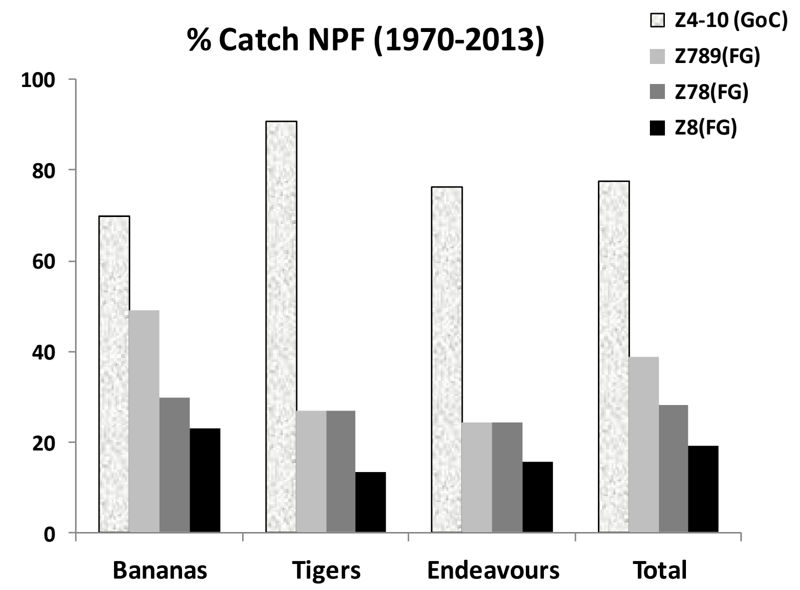


Figure 3.6 a&b (a) NPF fishing zones in the Gulf of Carpentaria (Zones 4 to 10). (b) NPF fishing zones adjacent to the Flinders and Gilbert river mouths used in assessments (Zone 8; Zone 7 to the west and 9 to the north). Log book data for prawns are collected on a 6nm (~11km) grid.

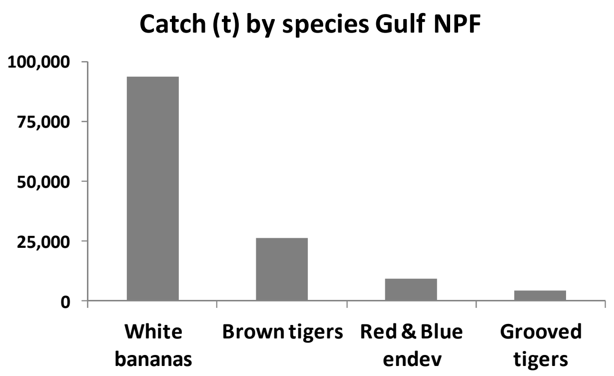
Woodhams et al. 2012). In the 2011 fishing season the NPF real value was estimated at $97.1M, and in 2012 $64.7M (Woodhams et al. 2012). Zones 4-10 (Figure 3.6a) encompass the GoC region of the NPF, and our assessment focuses on catch in Zones 7-9 in the southern GoC as they may be influenced more directly by reduced flows in the Flinders and Gilbert catchments, in particular Zone 8, which is directly opposite the EOS outflows into the coastal-marine receiving waters. Daily catch-effort data are recorded in log books in a 6nm (~11km) grid (Figure 3.6b). Fine spatial scale analysis of catch over the historical time series is beyond the scope of this project and is not attempted.

Across the whole NPF and all fishing seasons since 1970, the GoC accounted for 78% of the total catch, 70% of the Banana Prawn catch, 91% of the Tiger catch and 76% of the Endeavour catch (Figure 3.7a)..

(a)



(b)



(c)

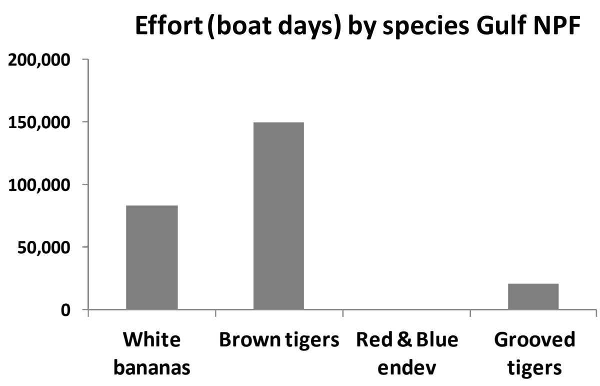


Figure 3.7 a-c. Comparison of the (a) percentage of total prawn catch by species for all zones in the NPF for: the Gulf of Carpentaria (zones 4 10); zones 7, 8 and 9 (southern and south-eastern GoC); zones 7 and 8 (the southern GoC) and Zone 8 opposite the Flinders and Gilbert catchments. Comparison of Banana Prawn (b) Catch (t) and (c) Effort (boat days) by prawn species caught in the GoC NPF fishing zones (4-10), 1970-2013.

White Banana and Tiger (Brown and Grooved combined) prawns were the dominant species caught in the GoC NPF by weight (t) between 1970 and 2013 (n=43y), with Banana Prawns comprising over three times the catch of Tigers Prawns (Figure 3.7a). In contrast, however, most effort (boat days) was expended catching Tiger Prawns, which was about twice that for Banana Prawns (Figure 3.7b). Both Red and Blue Endeavour Prawn species comprise a small proportion of catch and effort in the GoC and so are not assessed here. White Banana Prawns are therefore an ideal candidate for assessing potential impacts of reduced freshwater flow into the southern GoC because they are a significant component of the NPF catch with a known and strong positive correlation with rainfall-induced flow (Vance et al. 2003, 1991; Venables et al. 2011). Species descriptions for White Banana, Tiger and Endeavour Prawns are provided in Part 1 of this report (sections 8.3 and 8.4, respectively).

The total Banana and Tiger Prawn catch and effort in zones 7-9 over the NPF time series (1970-2013) show considerable small and large-scale variations, but have opposite underlying general trends (Figure 3.8a&b respectively). In general, as Banana Prawn effort and associated catch increased, that for Tiger Prawns decreased. The relationship between the two prawn fisheries in the NPF is complex; however the general

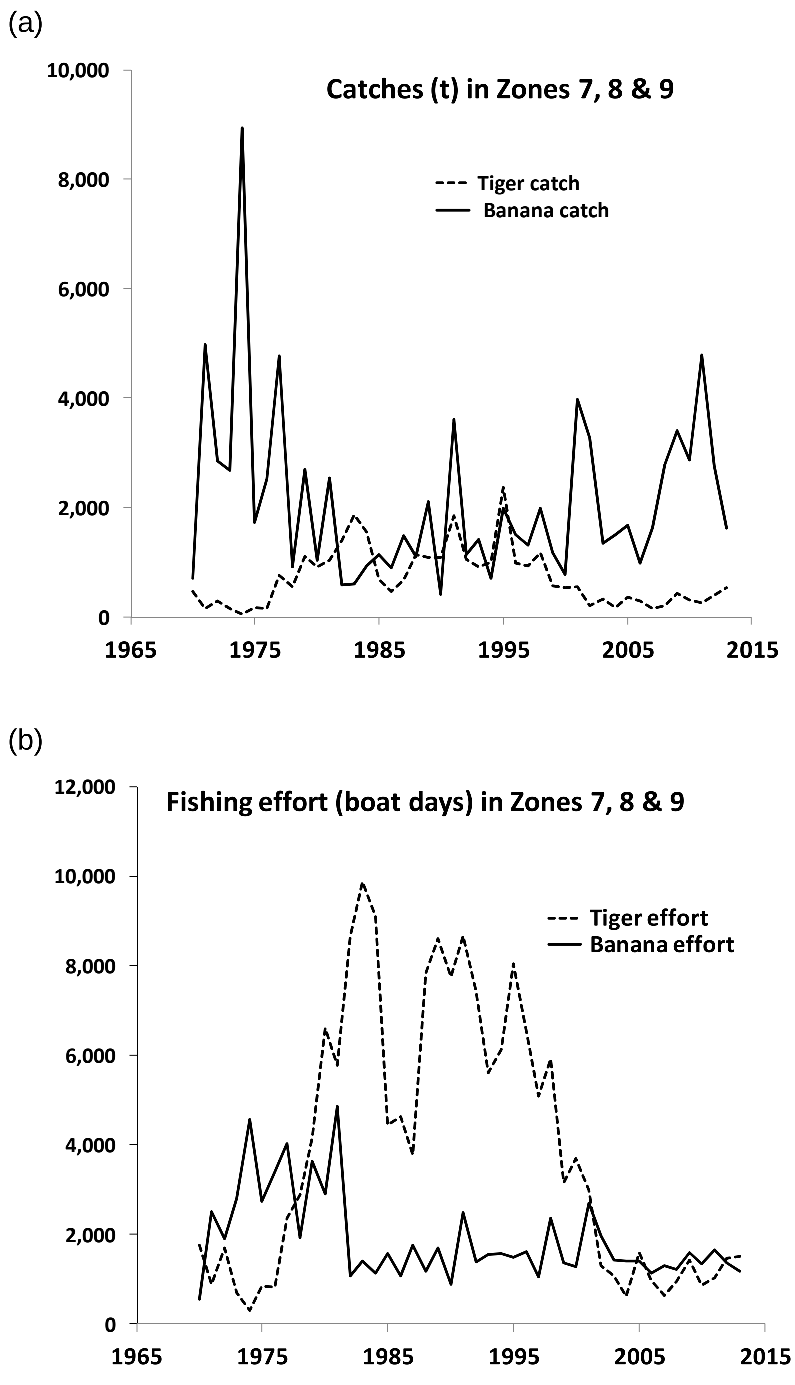


Figure 3.8 a&b. Trends in Banana and Tiger Prawn (a) catch (t) and effort (boat days) in Fishing zones 7, 8 and 9 in the Gulf, 1970-2013. Tiger Prawns includes both Grooved and Brown species combined.

trends suggest that variations in fisheries management and economic drivers (Dichmont et al. 2008, Pascoe et al. 2010, Kompas et al. 2010) are important drivers in explaining variations in catch as are environmental

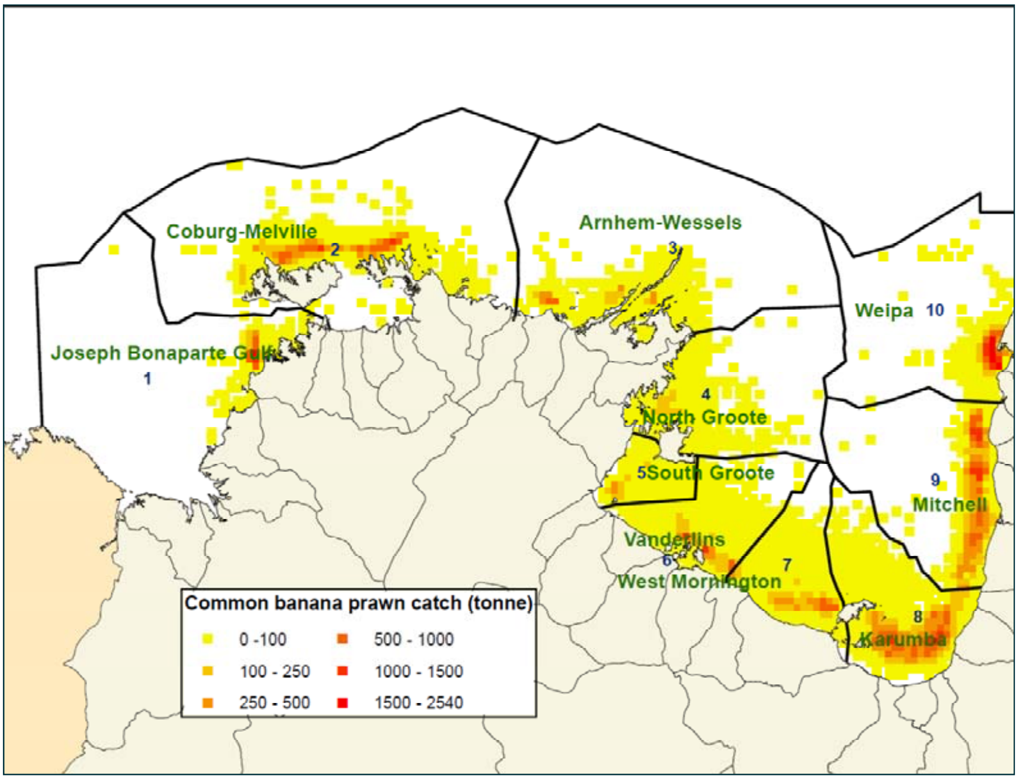


Figure 3.9 The spatial distribution of the total catch (1970-2013) of the Common Banana Prawn, The colour levels show the catch on a logarithmic scale, coastal sections of separate rainfall basins are shown in shades of grey (note: grids with < 100 t have been left out due to the commercial in confidence restrictions applied to logbook data and the 5-boat rule). Modified from Figure 1.3 in Venables et al. (2011).

conditions determining *in situ* abundance. Venables et al. (2006) provides a comprehensive analysis of log book data and methods used to ascertain species distribution and catch allocation in the NPF, recently enhanced by Zhou et al. (2014) by addressing catchability and fishing power increase with a focus on natural mortality. The relative distribution and abundance of Common Banana Prawns (both species combined) across the NPF are indexed by catch between 1970 and 2013 (Figure 3.9, Venables et al. 2011) showing that, in general, the southern and north-eastern Gulf is a 'hot spot' for Banana Prawn catch.

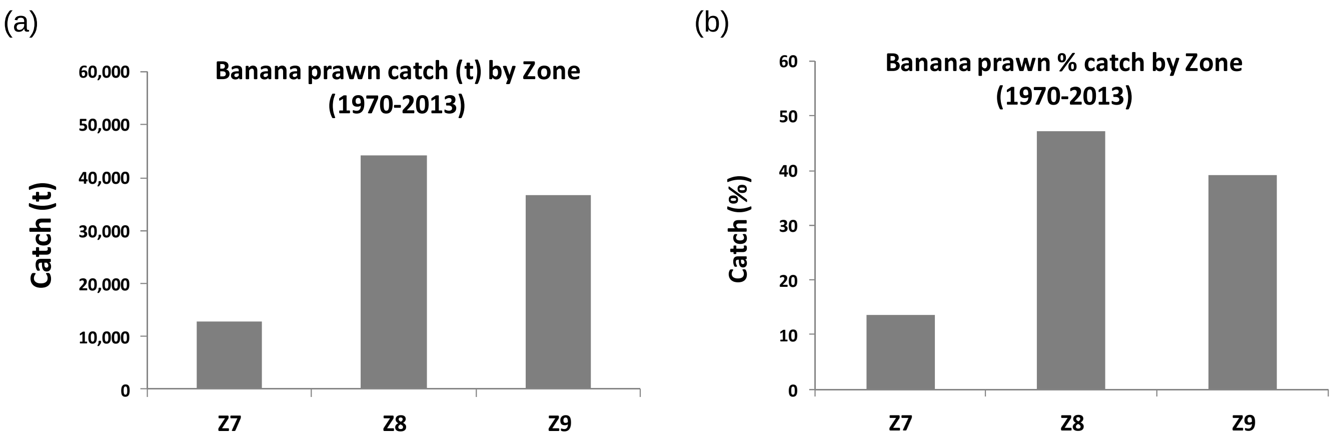


Figure 3.10 a&b Comparison of White Banana Prawn (a) catch by weight (t) and (b) % composition by weight between fishing zones 7, 8 and 9. Gulf of Carpentaria, 1970-2013. Zone 8 is opposite the Flinders and Gilbert river mouths, Zone 9 is to the north of the Gilbert catchment, and Zone 7 to the west of the Flinders catchment.

Of the three NPF fishing zones in the Gulf that were assessed in this study, Zone 8 had the highest total catch weight (t) between1970 and 2013 (Figure 3.10a) and, hence, percentage catch by weight (47%, Figure 3.10b), followed closely by Zone 9 (39%), with Zone 7 having considerable less (14%).

In summary, the above characterisation of the Gulf NPF since 1970 shows that in the marine and coastal receiving waters in fishing zones 7 and 8, adjacent to the Flinders and Gilbert catchments, the flow sensitive White Banana Prawn was the dominant catch by weight. However, the Tiger Prawns (Brown and Grooved) were dominant by effort (boat days).

12.1.2 RISK ASSESSMENT APPROACH

The risk assessment process is now applied to catchments and their aquatic ecosystems because it is transparent, consistent and reliable (Hart 2004, Hart et al. 2005, Chan et al. 2010). All steps in a risk assessment need to be guided at the outset by good conceptual models (Burgman 2005), and this caveat applies to both qualitative and quantitative methods (Bayliss et al. 2008, 2011). Conceptual models are abstractions of how we think the world works used to answer specific questions that may assist decision making, and usually takes the form of box-and-arrow diagrams (Drewery 2006).

The following generic approach to Quantitative Ecological Risk Assessment (QERA) outlined by Bayliss et al. (2008) for the Daly River catchment in the NT is adopted here: (i) construct a conceptual model that identifies hypothesised cause-effect links and interactions between assets and threats (here flow dependent species and their habitats from water extraction for FGARA agricultural scenarios, see Part 2); (ii) where adequate empirical data exist use frequentist statistics to characterise risk at a minimum and, if possible, develop more informative and predictive ecological models; (iii) make all uncertainties explicit and examine their influence on assessment outcomes using uncertainty and sensitivity analyses; and (iv) where there is combined reliance on empirical data and expert opinion/knowledge, and/or where decisions need to be made in the face of uncertainty, use Bayesian Belief Networks (BBNs) to capture uncertainties and communicate risk (Nayak and Kundu 2001; Chan et al. 2012; Bayliss et al. 2008, 2011; van Puten et al. 2013). The overall approach is similar to the risk assessment process proposed by Deere and Davidson (2005) for water management in Australia. However, due to time constraints it was not possible to develop BBNs with stakeholders or end users to communicate risks. The risk assessment approach used here is consistent with national and international guidelines with respect to robustness, transparency, coherency and reliability (e.g. see US EPA 1998, 2003; AS/NZS 2004a&b, AS/NZS ISO 31000 2009).

12.1.3 MODELS PREDICTING CATCH FROM FLOW AND EFFORT

The abundance of prawns can be highly variable because of the influence of regional and local environmental factors and broad-scale oceanographic and climate features such as cyclones (Vance et al. 2003). In particular, rainfall-induced river flow is highly correlated with offshore commercial catches of Banana Prawns in the south-eastern GoC (Vance et al. 2003). Increased river flow can have different effects on different stages of the White Banana Prawn life cycle: high flows can increase emigration of sub-adults from estuaries; increased flows can prevent immigration, settlement and survival of post-larvae; and rainfall-flow may increase the overall productivity through the contribution of increased nutrient input to increased growth and survival rates (see Part 2 Section 8.3; Staples and Vance 1986, 1987; Tanimoto et al. 2006; particularly Duggan 2012).

The statistical and risk modelling procedures developed by Bayliss et al. (2008) for the recreational Barramundi fishery in the Daly River (NT) to simulate potential risk from water extraction for agriculture are modified and applied to the GoC Banana Prawn fishery (and the Gulf Barramundi fishery, see Section 13). For NPF fishing zones 7, 8 and 9 empirical models were developed from observed catch and effort data, and Source model wet year EOS flow data for the Flinders and Gilbert rivers, to predict Banana Prawn catch between 1970 and 2013. The basic model is:

Catch (t) = constant + Effort + Flow ± model error

To assess the relative contributions of EOS flow from the Flinders and Gilbert catchments the preferred model is:

Catch Zone 8 (t) = constant + Effort + Flow (Flinders) + Flow (Gilbert) $\pm$ model error

Multiple regression analysis was used to test multiple hypotheses about the influence of X- independent variables chosen *a priori* (here fishing effort and flow) on the Y-dependent or response variable (the measurement endpoint in our quantitative risk assessment, catch weight). The performance criterion of the model is determined by the model R^2 (% explained variance in observed data) and its significance level, and the significance of each independent variable in the model.

Statistical analyses were undertaken in Statistica™ (StatSoft 2011) and all variables were examined for normality prior to analysis via normal probability plots, Kolomorov-Smirnov and Shapiro-Wilks' W tests. Y-X plots were used to examine for outliers, nonlinearity and uniform distribution of data across all levels. The relative influence of each independent variable is determined from their Beta coefficients (Zar 1974), which adjusts for different measurement units. Where appropriate $\log_{10}$ transformations were used to normalise ordinal data.

The effect of one independent variable may be influenced by the levels of other inter-correlated variables (here flow and effort) and, hence, there would be no single level of importance. Partial regression plots were used to examine the direction, magnitude and distribution of paired data in the multiple regression equations. These plots describe the effects of those variables on the response variable when the inter-correlated effects of all other variables are statistically held constant.

12.1.4 REDUCTION IN EOS FLOW FOR FGARA DEVELOPMENT SCENARIOS

The flow data comprised Scenario A, historical natural flow plus current water entitlements for the Flinders and Gilbert separately and combined. There was a small (up to 5%) difference between Source EOS data for natural historical flows and flows from Scenario A. For simulation purposes the amount of water lost to EOS flow from each FGARA development scenario, for the Flinders and Gilbert catchments, was assumed to be their annual yield allocations (i.e. full use of all water entitlements, Table 3.2). There would be a positive bias in this assumption because un-impounded water would be lost from the system before it reaches the river outlets, but the bias is expected to be small (<10%, C. Petheram pers. comm.). To arbitrarily increase the range of simulated water extractions we assumed the maximum harvest possible would be the maximum storage capacity for water. Additionally, we arbitrarily combined the B scenarios when combining the EOS flow for the two river systems. Whilst this may not be practically possible or desirable when assessing each of the six B scenarios, it simply serves the purpose of simulating potential loss of fisheries catch over a greater range of water harvests.

Table 3.2 FGARA Development Scenarios and quanta of water (GL) assumed to be used and unavailable to EOS flow and, hence, fisheries production.

NPF Zone & FGARA scenario	Combination of water harvests above current (scenario A)	Additional water harvest (GL)	% of mean WY Q	% of median WY Q
Flinders				
F Scenario B.1	Flinders B scenarios only	40	1	3
F Scenario B.2	Flinders B scenarios only	34	1	2
F Scenario B.3	Flinders B scenarios only	80	3	6
F Scenario B.1 (max)	Max = water storage capacity	248	9	17
F Scenario B.2 (max)	Max = water storage capacity	127	5	9
F Scenario B.3 (max)	Max = water storage capacity	560	20	39
Gilbert				
G Scenario B.1	Gilbert B scenarios only	172	5	7
G Scenario B.2	Gilbert B scenarios only	498	13	19
G Scenario B.3	Gilbert B scenarios only	17	0.4	1
G Scenario B.1 (max)	Max = water storage capacity	227	6	9
G Scenario B.2 (max)	Max = water storage capacity	725	19	28
G Scenario B.3 (max)	Max = water storage capacity	25	1	1
Flinders & Gilbert combined				
FG Scenario B.1	Add Flinders & Gilbert B scenarios	212	3	5
FG Scenario B.2	Add Flinders & Gilbert B scenarios	532	8	13
FG Scenario B.3	Add Flinders & Gilbert B scenarios	97	1	2
FG Scenario B.1 (max)	Max = water storage capacity	475	7	12
FG Scenario B.2 (max)	Max = water storage capacity	852	13	21
FG Scenario B.3 (max)	Max = water storage capacity	585	9	14

12.1.5 ACCOUNTING FOR VARIABILITY AND UNCERTAINTY IN DATA AND MODELS

Model simulations were undertaken in an Excel™-@Risk™ software environment (Palisade 2010). The conceptual model used to predict the impact of simulated wet season flow extractions on Banana Prawn (and Barramundi catch, see Section 13) in the Flinders and Gilbert rivers is illustrated in Figure 3.11, and shows all model uncertainties. The frequency distributions of the modelled wet year flow (log_{10}Q, ML) and fishing effort (log_{10}E, boat days) used in the multiple regression model to predict commercial catches (log_{10}Weight, t) were derived by using "Best Fit" in @Risk (2010) software.

Stochastic models were then developed that included the multiple regression error term and which replaced mean parameter estimates (the regression coefficients) with the pdf functions derived above (Figure 3.11). Monte Carlo simulations then accounted for uncertainty in the risk models predicting Banana Prawn catch on effort and flow. Monte Carlo methods are stochastic simulations that rely on repeated random sampling from a known distribution of data in order to estimate model parameters confidently, and are useful for modelling systems with significant uncertainty in inputs such as the calculation of risk (Hilborn and Mangel 1997).

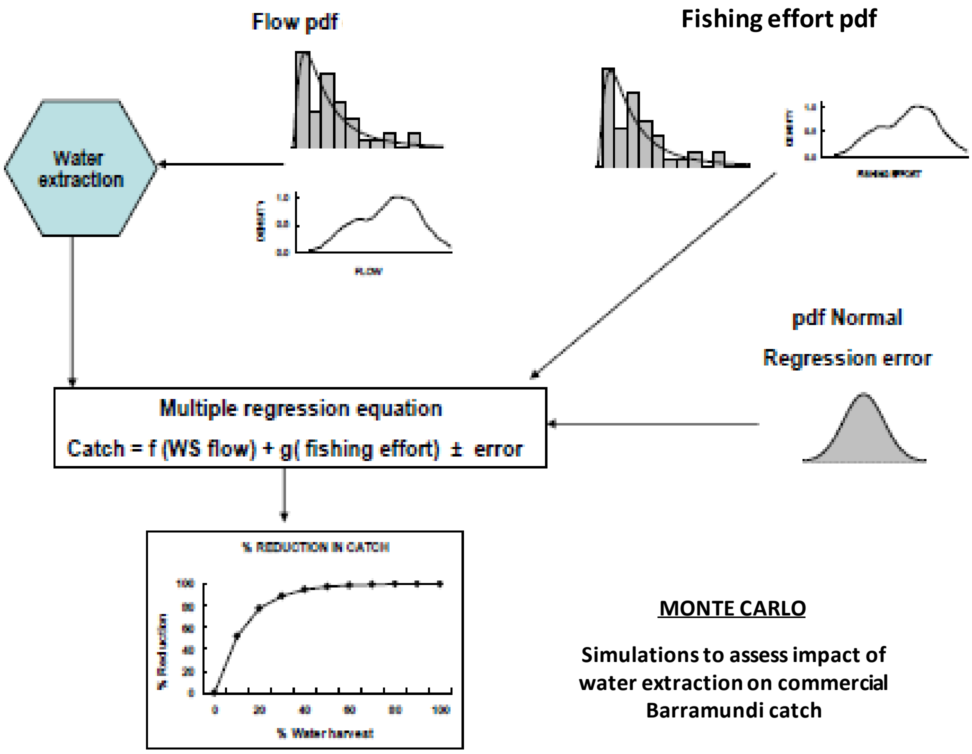


Figure 3.11 Conceptual model used to simulate the potential impact of water harvests under different FGARA agricultural development scenarios on two key fisheries species in the southern Gulf of Carpentaria (White Banana Prawns and Barramundi). From Bayliss et al. (2008).

The importance of parameter inputs on risk outputs was examined using sensitivity analysis of all outputs. All pdfs were randomly sampled by Monte Carlo simulation 10,000 times or more to derive a stable mean value. @Risk™ simulation results include graphical displays of the distribution of all results from outputs (e.g. via frequency and cumulative probability distributions), and generates sensitivity and scenario reports that help identify those inputs that are most critical to outputs. Sensitivity analysis is undertaken using regression analysis, whereby sampled input variable values are regressed against output values leading to a measurement of sensitivity by input variable. Results of the sensitivity analysis are displayed as a 'Tornado' type chart, with longer bars at the top representing the most significant input variables in a positive or negative direction (Palisade 2010). Results can also be displayed as a Box-and-Whisker chart, which is used here to display the risk to fisheries of each FGARA development scenario (e.g. the distribution of the simulation runs).

12.2 Results - Banana prawns and flow

12.2.1 MODEL SELECTION

In all zones fishing effort explained most of the variation in observed catches (Model 1: 46-72% R^2, Table 3.3), with the correlation on flow being strongest in Zone 8 opposite the Flinders and Gilbert catchments (Model 2: R^2= 43%, Table 3.3). Zone 9 had the weakest correlation with both effort and flow (Table 3.3). This large zone lies north of the Flinders and Gilbert catchments and was not expected to have a strong correlation with their flows apart from the Gilbert (Model 2: R^2 = 12%). Whilst Model 3 is the preferred model because it would allow assessment of the relatively independent contributions of EOS flow to catch in all zones, particularly zone 8, flow from the Flinders was a non-significant entry in the equation with most variation in catch explained by flow from the Gilbert. Catch in all zones were mostly explained by effort and flow from the Gilbert (Model 4: R^2 = 56 – 80%) compared to the Flinders (Model 5: R^2=48-62%), although the two flow series are strongly correlated. Given that we do not know the contributions of the Flinders and Gilbert flows to the total amount of freshwater entering zones 7 and 8 relative to flows from other nearby catchments, zones 7 and 8 were combined and analysed separately, although with similar results (Table 3.3).

Table 3.3 Summary of the multiple regression models used to predict Banana Prawn catch (t) in fishing zones 7, 8 and 9 (and zones 7 and 8 combined) as a function of fishing effort (boat days) and wet year flow (Q, ML) for the Flinders (FQ) and Gilbert (GQ) rivers, and for their combined flow (FGQ).

Model		Zone 7	Zone 8	Zone 9	Zone 7 & 8
1. Effort (E, boat days)		72%	55%	46%	71%
2. Flow (WYQ, ML)					
	Gilbert GQ	35%	43%	12%	41%
	Flinders FQ	24%	24%	ns	27%
	Combined Flinders FGQ	31%	42%	10%	38%
3. Catch (t) = E + FQ + GQ		78% (FQ ns)	70% (FQ ns)	55% (FQ ns)	80% (FQ ns)
4. Catch (t) = E + GQ		80%	70%	56%	80%
5. Catch (t) = E + FQ		75%	62%	48% (ns)	75%
6. Catch (t) = E + FGQ		77%	70%	53%	79%
Variation "explained" by flow not explained by effort		5%	15%	10%	8%

All data (1970-2010) were transformed to $\log_{10}$. Unless stated otherwise (ns), all regression models and variables are significant

Model 6, which combines the flow from the Flinders and Gilbert rivers, was chosen for assessment of reduced flow on catch because it had the highest variability in observed catch in Zone 8 explained by flow (15%, Table 3.3) that was not explained by effort in the joint regression equation. Zone 9 used only the Gilbert flow in assessment. The single regression relationships between flow and effort for Model 6 in zones 7-9 are illustrated in Figure 3.12a-f, respectively.

The multiple regressions equations used to predict Banana Prawn catch from fishing effort and wet year flow (1970-2011) are summarised in Table 3.4a-c for zones 7, 8 and 9 respectively. Effort had much more influence on observed catch than flow in zones 7 and 9 (2.9 and 2.6 times, respectively), and, in contrast, effort only explained 1.5 times the variation in catch than flow in Zone 8.

Table 3.4 a-c. Summary of the multiple regressions equations used to predict Banana Prawn catch ($\log_{10}$W, t) from fishing effort ($\log_{10}$E, boat days) and wet year flow ($\log_{10}$QFG, ML) in zones 7, 8 and 9 respectively (1970-2011). QFG is combined Flinders and Gilbert flow, QG is Gilbert flow.

(a) Zone 7: R= 0.885, adjusted R^2 = 77%, n= 42, P< 0.001, SE (standard error) regression = 0.286

Variable	Beta	SE Beta	B	SE B	P
Intercept			-1.785	0.688	<0.01
QFG (Flow)	0.259	0.082	0.341	0.111	<0.001
E (Effort)	0.747	0.082	0.756	0.084	<0.001

(b) Zone 8: R= 0.834, adjusted R^2 = 70%, n= 42, P< 0.001, SE regression = 0.207

Variable	**Beta**	**SE Beta**	**B**	**SE B**	**P**
Intercept			-1.755	0.509	<0.001
QFG (Flow)	0.395	0.096	0.834	0.136	<0.001
E (Effort)	0.589	0.096	0.335	0.082	<0.001

(c) Zone 9: R= 0.743 adjusted R^2 = 53%, n= 42, P< 0.001, SE regression = 0.228

Variable	**Beta**	**SE Beta**	**B**	**SE B**	**P**
Intercept			-1.016	0.635	=0.118
QG (Flow)	0.256	0.108	0.191	0.081	=0.02
E (Effort)	0.662	0.108	0.890	0.146	<0.001

The partial residuals plots of Model 6 multiple regressions equations predicting Banana Prawn catch from fishing effort and the combined Flinders and Gilbert wet year flow for zones 7 and 8 are illustrated in Figure 3.13a-d. These plots show the influence of each variable in the joint equation when the influence of the other variable is held constant at its mean value. The regression equation for Zone 9 uses Gilbert River flows only (Figure 3.13e&f). Results show that the effect of flow on catch in each zone is enhanced when the effect of effort is factored out.

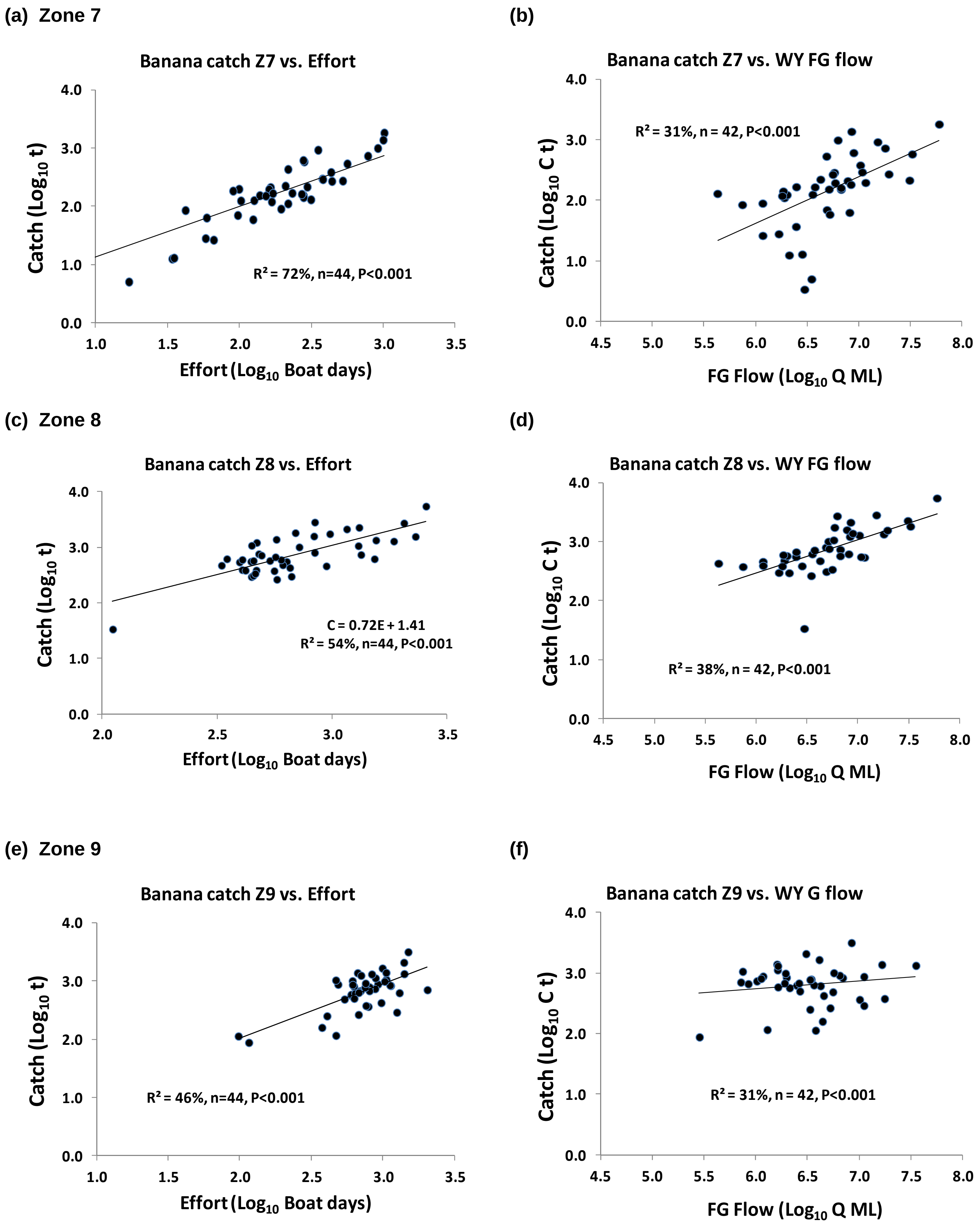
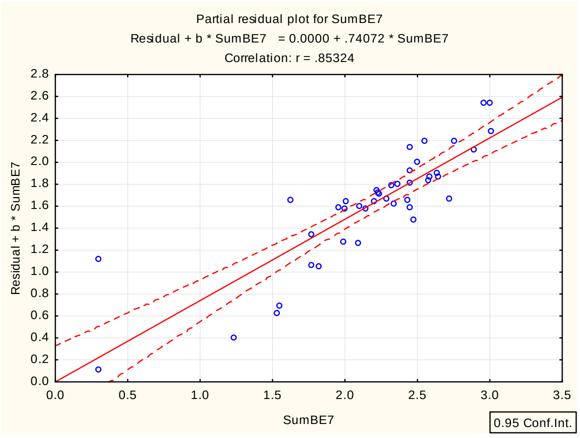
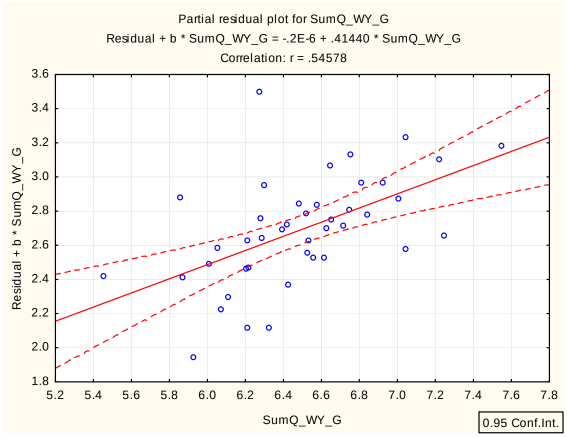


Figure 3.12 a-f Regression between Banana Prawn catch ($\log_{10}$C, t) on fishing effort ($\log_{10}$E, boat days) and wet year flow volume ($\log_{10}$Q, ML) for zones 7 (a & b), 8 (c & d) and 9 (e & f), respectively. Gulf NPF (270-2011).

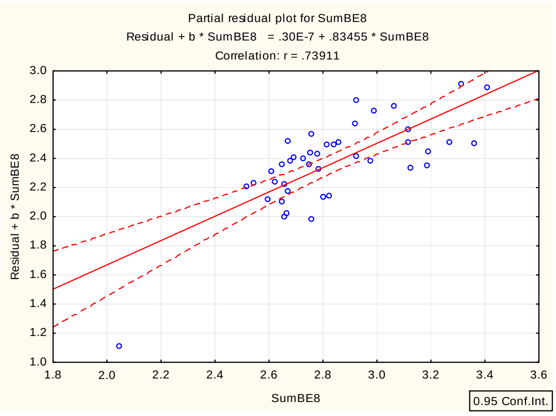
(a) Zone 7 Effort



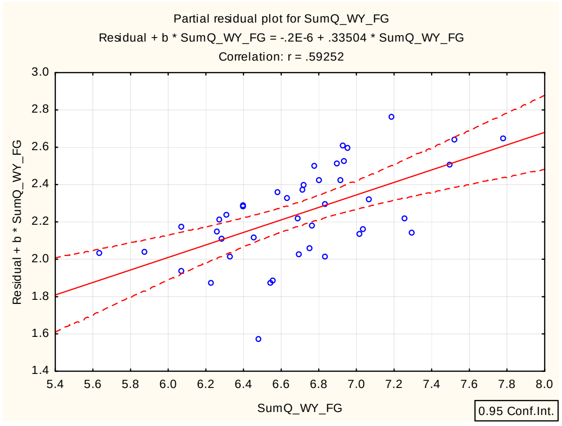
(b) Flow



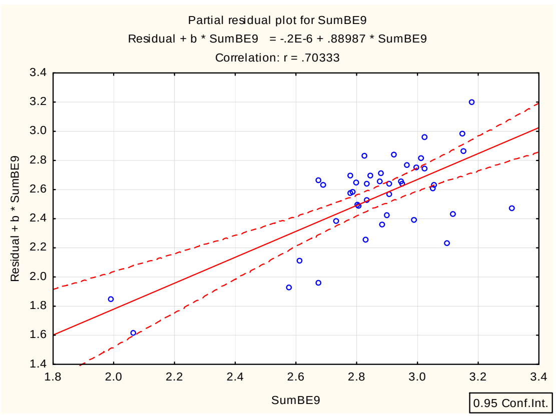
(c) Zone 8



(d)



(e) Zone 9



(f)

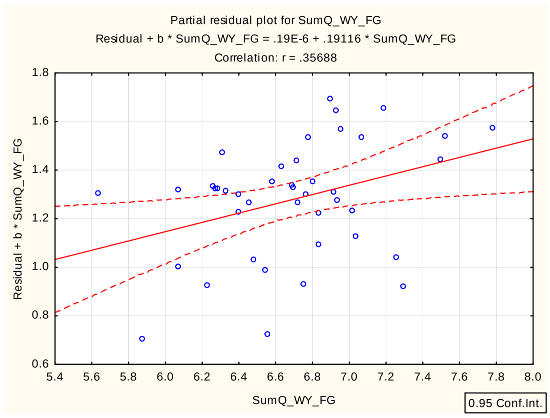


Figure 3.13 a-f Partial residuals plots of the multiple regression equations predicting Banana Prawn catch (t) from effort (boat days) and flow (wet year, ML) for zones 7 (a & b respectively), 8 (c & d respectively) and 9 (e & f respectively), 1970 – 2011. Catch in zones 7 and 8 were correlated to the combined Flinders and Gilbert flows, and for Zone 9 only to the Gilbert flow. All data were transformed to $\log_{10}$.

12.2.2 PREDICTED REDUCTION IN CATCH FOR FGARA DEVELOPMENT SCENARIOS

The equations derived above were used to predict the reduction (%) in the mean 1970-2013 Banana prawn catch in zones 7-9 for each FGARA development scenario (i.e. water harvest from annual yield to maximum storage capacity, see Table 3.2). For simulation purposes it is assumed that all scenarios between catchments are not mutually exclusive and can occur in any combination. The predicted reduction in catch in Zone 7 is 2% or less for all B scenarios combined across the Flinders and Gilbert catchments (Figure 3.14a, grey bars). Where we assume that maximum storage capacity is used instead of annual yield, the predicted reduction in catch is 3% or less (Figure 3.14a, black bars). The predicted reduction in catch in Zone 8 is 4% or less for all B scenarios combined across the Flinders and Gilbert catchments (Figure 3.14b, grey bars). Where we assume maximum storage capacity the predicted reduction in catch is 7% or less (Figure 3.14b, black bars). The predicted reduction in catch in Zone 9 is 3% or less for all B scenarios across the Gilbert catchment (Figure 3.14c, grey bars). At maximum storage the predicted reduction in catch is 5% or less (Figure 3.14c, black bars). The predicted reduction in catch across all catch zones is 7% or less for all B scenarios (Figure 3.14d, grey bars). Where we assumed maximum storage the predicted reduction in catch is 11% or less (Figure 3.14d, black bars).

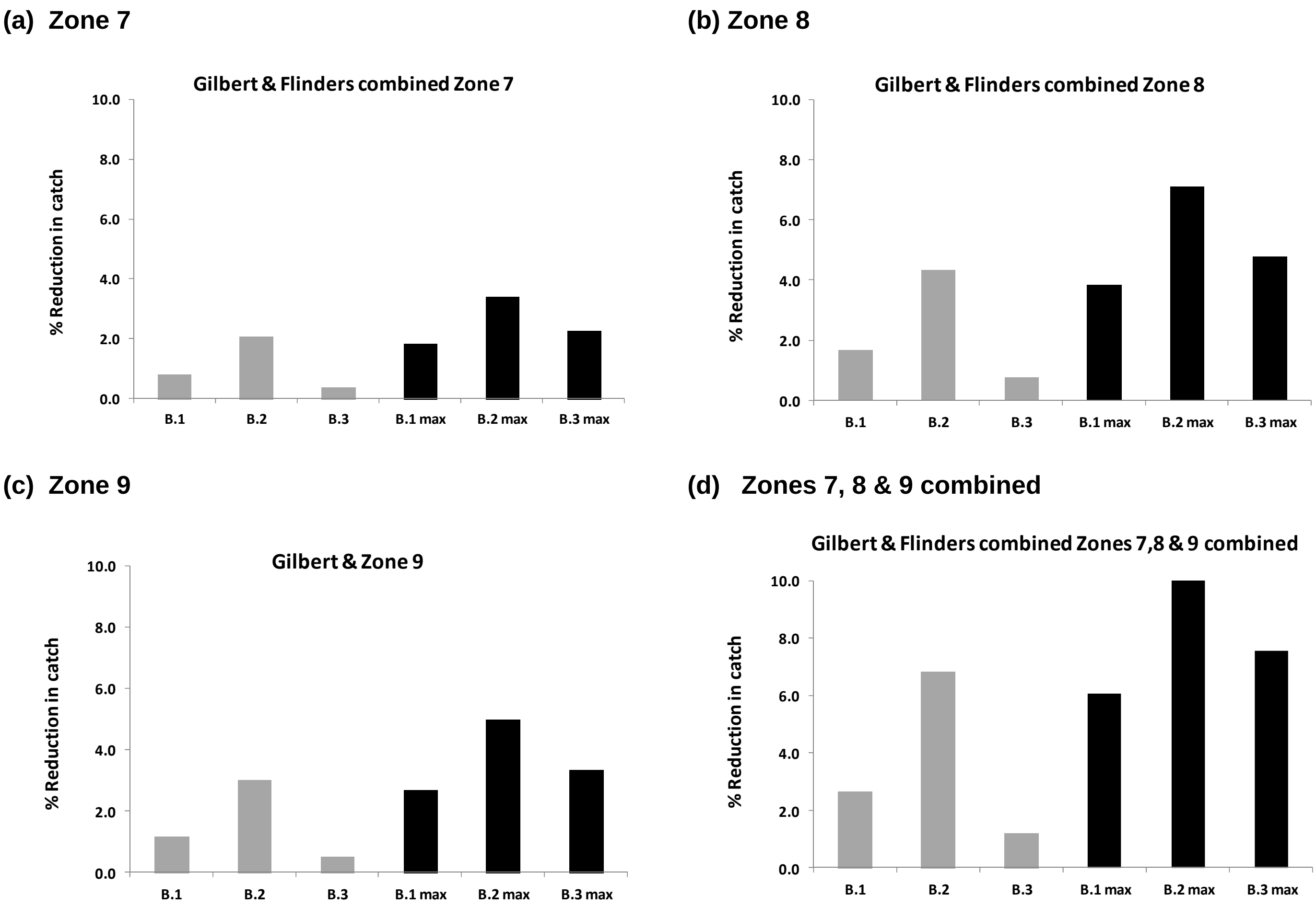


Figure 3.14 a-d. Predicted percentage (%) reduction in the mean (1970-2013) Banana Prawn catch in zones (a) 7 and (b) 8, for the Flinders and Gilbert EOS flow combined, for each FGARA development scenario (i.e. water harvest from annual yield (grey) to maximum storage capacity (black), see Table 3.4).

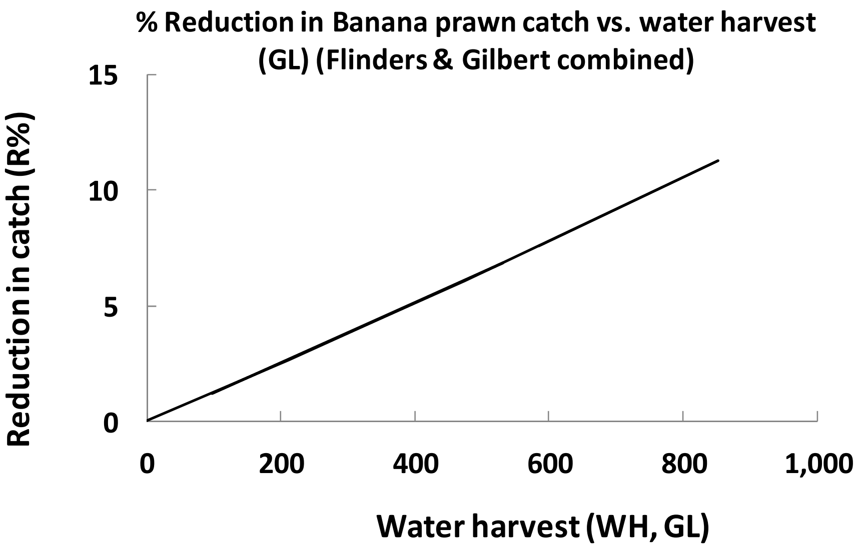


Figure 3.15 Predicted reduction (%) in mean (1970-2013) Banana Prawn catch (t) from reductions in EOS flows (GL) due to the water needs of each FGARA development scenario (ranging from predicted annual yield to maximum capacity of storages). For Zone 8, the Flinders and Gilbert development scenarios (B) were arbitrarily combined.

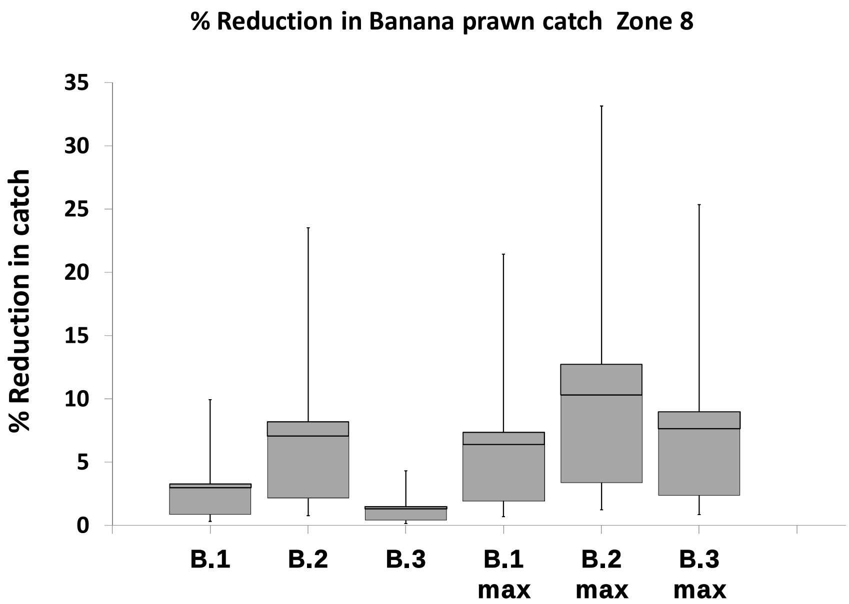


Figure 3.16 Predicted reduction (%) in mean (1970-2013) Banana Prawn catch in Zone 8 for each FGARA development scenario when uncertainty in models are included. For Zone 8 the Flinders and Gilbert scenarios were arbitrarily combined, along with their EOS flows. Grey boxes encompass values for 50% of all simulations (25% to 75%), the dark horizontal lines are mean values and the vertical lines represent the 5% to 95% range. For Zone 8, the Flinders and Gilbert development scenarios (B) were arbitrarily combined.

The simplistic relationship between water harvest and the predicted percentage reduction in Banana Prawn catch in Zone 8 is plotted in Figure 3.15. However, this does not account for model error (including Source model errors) or intrinsic variability in effort and flow, and nor does it account for periods of low-flow conditions or prolonged droughts. The results of the uncertainty analysis for Zone 8 are illustrated in Figure 3.16 and give a range of estimates for predicted reductions in catch for each B scenario (after 10,000 Monte Carlo simulations). For the combined Flinders and Gilbert B.2 scenarios, 50% of simulations predicted a 2-8% (mean 7%) reduction in mean Banana Prawn catch. Where we assumed maximum storage is used a 3-13% (mean 11%) reduction in mean catch is predicted.

12.3 Discussion and Recommendations

Part 2 of this assessment report rated the risk to the Banana Prawn fishery in the south-eastern corner of the GoC (Zone 8) from the FGARA development scenarios as high, and this is not surprising given the substantial body of research over several decades to support the strong relationship between their commercial catch and river flow. Life history studies (e.g. Staples 1984; Staples and Vance 1986, 1987; Vance 1991; Vance et al. 2003), coupled with a sophisticated predictive statistical model of catch on regional rainfall (Venables et al. 2011; Buckworth et al. 2014), suggest strongly that the consequences, or population-level effects, of reduced flow would pose significant risk to catch and, hence, economic return to the NPF.

The advantages of a rainfall-catch model are substantial given that regional rainfall data are readily available and correlated closely to regional river flow, which itself is correlated closely to hydro-ecological processes that affect population level responses of Banana Prawns and, ultimately, their abundance and/or catchability in the GoC NPF. The rainfall-catch model has great management application also in that catch can be anticipated ahead of time and fishing controls fine-tuned accordingly (peak rainfall-flow is Jan-March and the fishing season April-June). However, this has been against the background of little to no large-scale agricultural developments and associated reductions in EOS flow that support fisheries production. There is now an increasing need to move to NPF assessment models that use estimates of EOS flow at catchment scale rather than observed rainfall at regional scale given the increasing pressure for agriculture development in the north. This may be possible for the Flinders and Gilbert catchments using Source model outputs developed for FGARA.

The objective of this quantitative assessment was to use available data to gain a better understanding of the likelihood of exposure to risk by attempting to quantify the magnitude of the potential reductions in catch associated with FGARA water harvest scenarios. Statistical models were therefore developed using historical data to predict Banana Prawn catch in the southern GoC from fishing effort and the combined Flinders and Gilbert river flow. Not surprisingly, the flow-effort-catch models explained similar amounts of variability in annual catch (~70%) for Zone 8 as that for the rainfall-catch model over larger regional scales (the whole NPF) and where catch is adjusted for effort.

The statistical models for zones 7-9 were then used to simulate 'what if' scenarios that cover a range of water harvests associated with the FGARA development scenarios. It was not possible to statistically separate out the independent effects of EOS flow from each catchment on Banana Prawn catch in Zone 8 given their strong cross-correlation. Hence, their flows were combined necessitating the creation of 'new' artificial B development scenarios (e.g. B.1 Flinders is combined with B.1 Gilbert and so on). Model and parameter uncertainties were incorporated into all risk analyses using Monte Carlo simulations and sensitivity analyses, providing a range of estimated reductions in catch rather than a mean value. Given small sample sizes model predictions were not validated with independent data or by using Bootstrapping re-sampling methods (Wu 1986) on the same data.

Overall the simulated risk results suggest that water harvests at the level of FGARA scenarios will likely reduce Banana Prawn catch in Zone 8 by 2 to 8%. If maximum storage is used then this increases the range to 3 to 13%. The maximum extraction rate of water in the Flinders is the B.3 scenario at 21% of mean EOS wet year flow and 41% of the median value. For the Gilbert, the maximum extraction rate is for the B.2 scenario at 20% of mean EOS wet year flow and 28% of the median value.

There are two major issues that may confound these model results. The first is combining the Flinders and Gilbert flows in order to predict catch in Zone 8. This seemed practical given that the distance between river mouths is only about 70 km. A simple spreadsheet model shows that a 2-catchment catch-flow model will give the same result as a 1-combined catch-flow model if the relationship between catch and per unit of flow is the same irrespective of its source (and is a reasonable assumption), but only if no other flow inputs are allowed. Reduction of flow in either catchment model doesn't affect this outcome. However, the addition of a third catchment input of say equal flow magnitude as the other two will overestimate the catch attributed to the first two by one-third. This would be analogous to flow inputs from the Norman River into Zone 8 as it is situated between the Flinders and Gilbert rivers. Hence, our results likely overestimate catch attributed to the Flinders and Gilbert rivers to an extent proportional to the EOS flow of the Norman River and other surrounding rivers such the Staaten River to the north and the Leichhardt and Nicholson-Gregory rivers to the west, particularly given the slow clockwise mean circulation in the GoC that appears to be a permanent feature (Forbes and Church 1983, Forbes 1984). Nevertheless, spreadsheet modelling shows also that if the catch-flow model is assumed the same irrespective of source then the relative reduction in catch with a reduction in flow should be the same. However, these are assumptions that require detailed testing by modelling the potential contributions of all sources of EOS flow to Banana Prawn catch in Zone 8 and, additionally, zones 7 and 9. Preliminary analysis of the relative contributions of EOS flow from catchments flowing into NPF zones 7, 8 and 9 are: Flinders 16%; Gilbert 24%; Leichhardt 11%; Norman 22% and Staaten 26%. The Flinders and Gilbert combined contributes to 41% of EOS flow to NPF zones 7, 8 and 9. An IQQM model was used to estimate EOS flow for the Leichhardt River (1930-2011), Source models for the Flinders and Gilbert (1890-2010), and NASY rainfall-runoff models for all others (1930-2011)(Petheram et al. 2009a,b).

The risk models reported here are deceptively simplistic in that they assume there are no other factors affecting catch of Banana Prawns other than fishing effort and flow. Future assessments should consider the cumulative effects of other multiple risks to Banana Prawn catch to provide a more realistic context. For example, Dichmont et al. (2001) analysed risk to the sustainability of NPF stocks from a fisheries management perspective, Welch et al. (2014) assessed the risks to the NPF from climate change, and Zeller and Snape (2006) undertook an Ecological Risk Assessment of Queensland-Managed Fisheries in the GoC from the perspective of their impacts on biodiversity.

Tiger prawns have a life history that is not directly dependent on freshwater flow and, hence, they have been qualitatively assessed in Part 2 as being at no risk from reduced flow due to large-scale agricultural development. The results of our quantitative assessment support this assessment apart from one anomalous combination. Although Tiger Prawn catch in Zone 7 was mostly explained by effort (R^2 = 92%, n=44, P<0.001), the addition of flow from the Flinders River was a significant entry into a multiple regression equation, albeit only explaining an extra 2% of variation in observed catch. Whilst effort had 10 times the influence on catch than flow, this result should be examined in more detail. Tiger Prawns depend on sea grass habitat, whose productivity may be influenced by nutrients discharged with flow. However, Burford et al. (2009) argued that, on a whole of GoC basis, river nitrogen inputs are unlikely to be major contributors to primary productivity.

12.3.1 RECOMMENDATIONS

i. Finer spatial (6nm grids) analysis of catch data and effort is required in order to determine the relative contributions of the Flinders, Norman and Gilbert rivers EOS flow to Banana Prawn catch in Zones 7, 8 and 9, particularly Zone 8.

ii. The contributions to total flow in the GoC of all major catchments needs to be determined using NASY rainfall runoff models to place the results reported here into better regional context (e.g. the Staaten and Mitchell to the north of the Gilbert, and the Leichhardt, Nicholson and other western GoC catchments including the Roper to the west of the Flinders).

iii. Further consultation and engagement with the NPF industry is required to communicate these risks accurately with respect to the true uncertainties in the risk models. The risk simulations are simply "what if" scenarios and are underpinned by many untested assumptions in the predictive catch-effort-flow models used here.

iv. Other risk-based decision support tools, such as Bayesian Belief Networks (see Cain 2001), should be developed to incorporate the range of expert opinions normally associated with perceived risks, and to assess, evaluate and then communicate the effectiveness of mitigation strategies proposed in Part 4 of this report.

13 Coastal Gulf fisheries (Barramundi)

13.1 Methods

13.1.1 CHARACTERISATION OF THE FISHERY

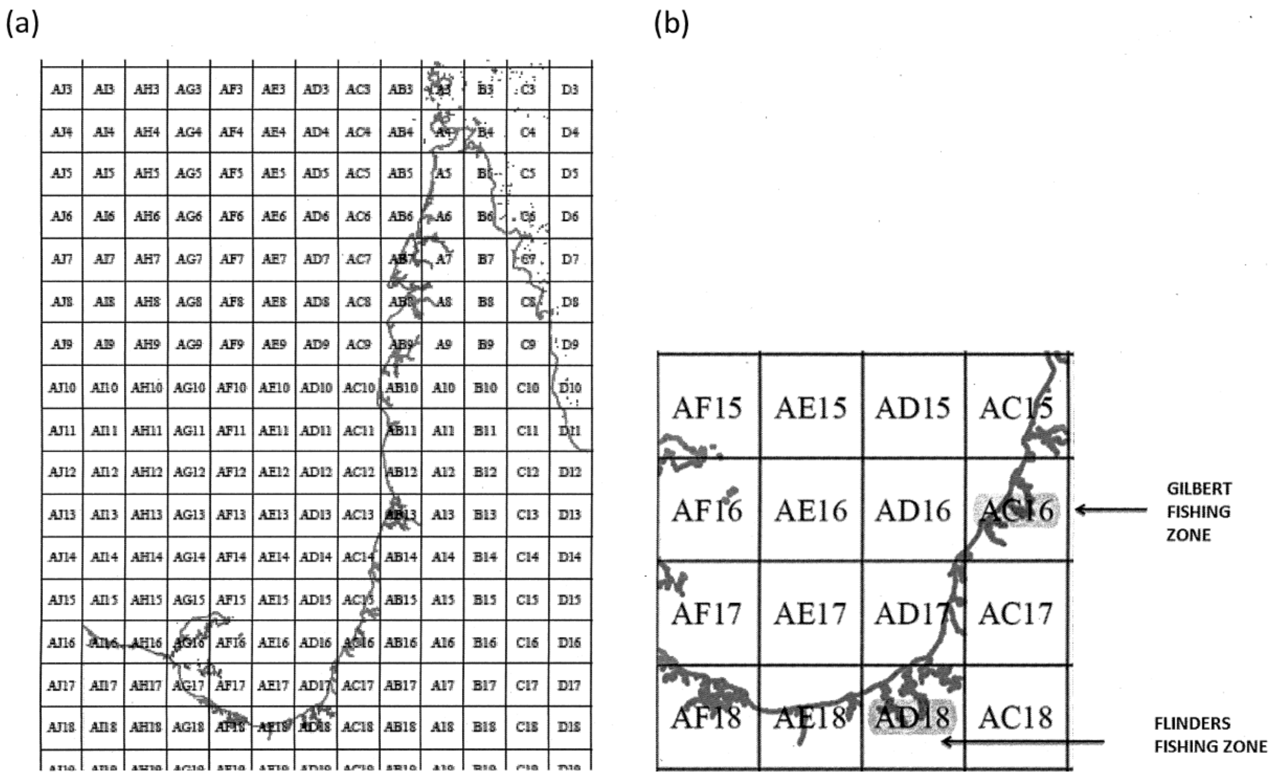
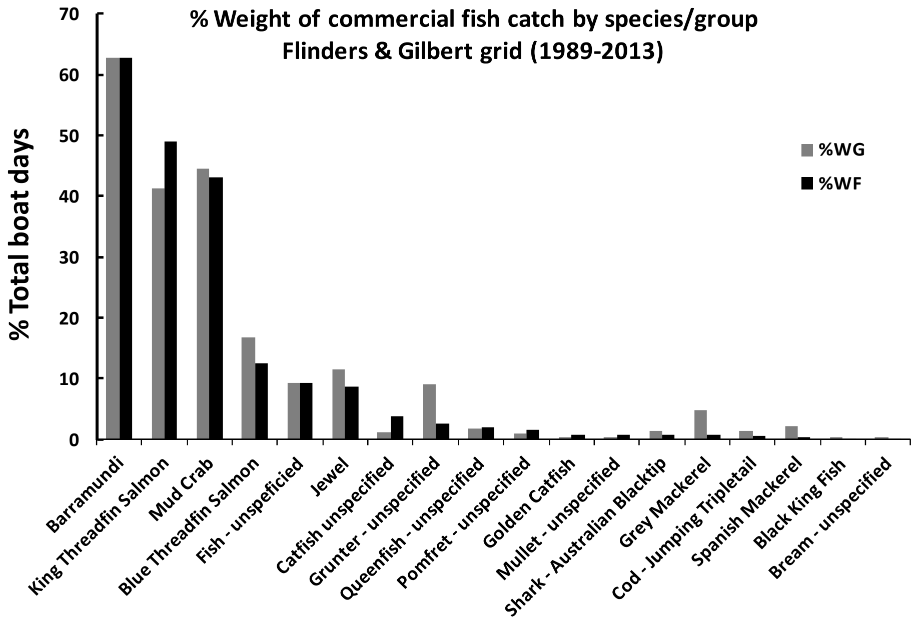


Figure 3.17 a&b. (a) Queensland coastal fishing log book grids (zones) in the Gulf of Carpentaria, and (b) the Gilbert (AC16) and Flinders (AD18) reporting grids (data from DAFF). Each grid is approximately 30nm encompassing a finer resolution 6nm grids (Robins pers. comm.).

A description of the Queensland Gulf of Carpentaria Inshore Finfish Fishery (GOCIFF) is provided in Part 2 sections 7.3 and 7.5. Catch (t) and effort data (boat days for nets, lines and pots) for all species caught in the commercial GoC fishing zone (here called QGF and includes the Mud Crab fishery) were obtained from Fisheries Queensland (DAFF). Log book data were collected on 30nm grids illustrated in Figure Figure 3.17a, and for the Flinders and Gilbert rivers Figure Figure 3.17b. The catch recorded between 1989 and June 2013 in grid AC16 is from the Gilbert River, whereas the catch recorded in grid (AD18) comes from the both the Flinders, Norman and Bynoe rivers, with the most of the catch from the Flinders River.

Of catch data recorded to species (n=11) Barramundi comprised the most of the catch by weight (Figure 3.18a) and entailed most of the fishing effort (Figure 3.18b) in both the Flinders and Gilbert rivers, followed by King Threadfin (Polydactylus macrochir) and Mud Crabs (Scylla serrata). Hence, although our assessment is on Barramundi, both latter species are examined for flow relationships to provide better context for the importance of flow to this fishery.

(a)



(b)

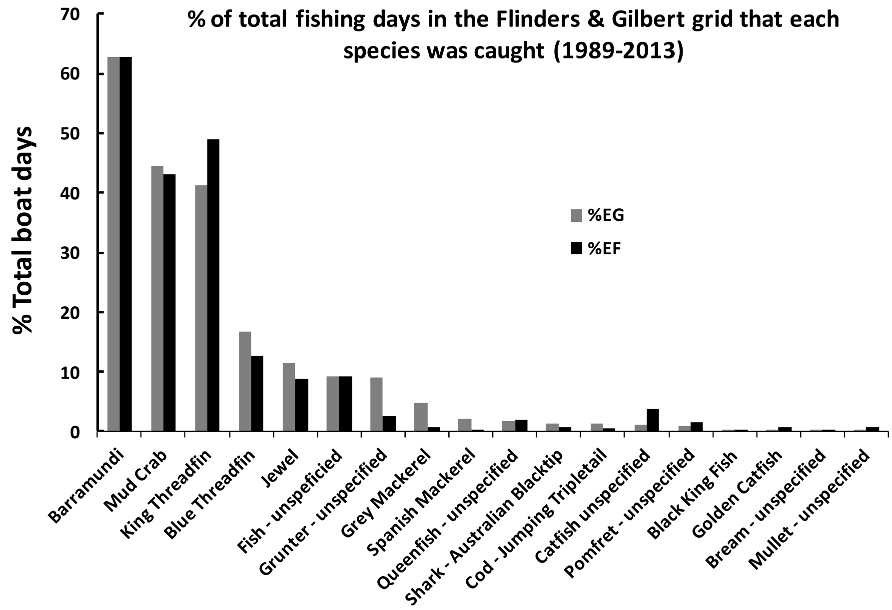


Figure 3.18 a&b (a) Comparison of the % composition by weight of commercial fish catch by species in the Flinders and Gilbert rivers (1989-2013), and (b) that for fishing effort (% of total fishing days in each grid that each species was caught).

In contrast to the Banana Prawn fishery (season = April-June), Barramundi fishery catch and effort in both grids were not consistent by month throughout the year. Hence, CPUE (catch per unit effort, weight/boat days) indices were derived to standardise for variable effort in order to characterise their relationships with Wet Year flow (ML) over the same time interval (October to September). Both CPUE indices and flow data were transformed to $\log_{10}$. Autocorrelation was undertaken on each of the three species in both rivers over the 22y data set, and shows interesting patterns although not consistent between rivers. Barramundi CPUE in the Gilbert River were negatively correlated with an approximate 10y time lag (Figure 3.19a), suggesting an 'average' return period of 30y that is consistent with the decadal patterns of combined Flinders and Gilbert flow described in Section 11.2.1 (Figure 3.5 a&b). Mud Crabs caught in the Gilbert show a similar pattern (Fig. 3.19e). In contrast, there were no strong return periods for King Threadfin (Figure 3.19c). All three species in the Gilbert show strong positive cross-correlation with wet year flow with no time lags (Figure 3.19b, d & f). Barramundi CPUE show no significant correlation with lagged flow (Figure 3.19b), which is in contrast to the results of Bayliss et al. (2008) for the Daly River (NT) who found strong positive correlations at 0-3y time lags. This result suggests that the correlation with instantaneous flow may be confounded with increased movements during the wet season, as a 2-3y time lag will likely reflect the time it takes for a juvenile Barramundi to grow and reach the legal fishery size limit. Similar time series results were found for these three species in the Flinders, except that King Threadfin was not cross-correlated with flow.

Other metrics associated with river flow were examined also for cross correlation with CPUE indices of the three species (see Table 3.5, results only for the Flinders River). These were: the maximum wet year flow and the mean; and mean monthly minimum and maximum area of floodplain flooded per wet year. The latter three metrics may index the availability of floodplain nursery habitat for Barramundi and connectivity to the river. However, apart from both minimum values, these metrics were highly correlated with total wet year Flow (Table 3.5). Barramundi CPUE in the Flinders River had a stronger positive correlation with minimum wet year flow than for total wet year Flow. This result should be investigated further as it may indicate a threshold flow value for water management purposes.

Correlations between CPUE and dry season flow (May to October), early wet (October to December), mid wet (January to March) and late wet (April & May) were examined also to check for seasonal patterns. However, in all cases, the strongest positive correlations were found with mid-wet season flows.

Table 3.5 Correlation matrix between wet year flow metrics that may influence Barramundi life history differently to total flow volume.

Variable	WY Flow	Min WY Flow	Max WY Flow	Mean Area	Min Mean Area	Max Mean Area
WY Flow	1.000					
Min WY Flow	**0.449**	1.000				
Max WY Flow	**0.972**	0.369	1.000			
Mean Area	**0.971**	**0.569**	**0.914**	1.000		
Min Mean Area	**0.447**	**0.988**	0.368	**0.560**	1.000	
Max Mean Area	**0.966**	**0.414**	**0.974**	**0.920**	**0.420**	1.000

WY Flow = wet year flow volume (ML), Min WY Flow = the minimum flow volume in any wet year (ML), Max WY Flow = the maximum flow volume in any wet year, Mean Area = the mean monthly mean daily area (km^2) of floodplains flooded in any wet year (here the Flinders), Min Mean Area = the minimum area value in any wet year and Max Area = the maximum area value in any wet year. Correlation coefficients highlighted bold and with larger font are significant at P<0.05.

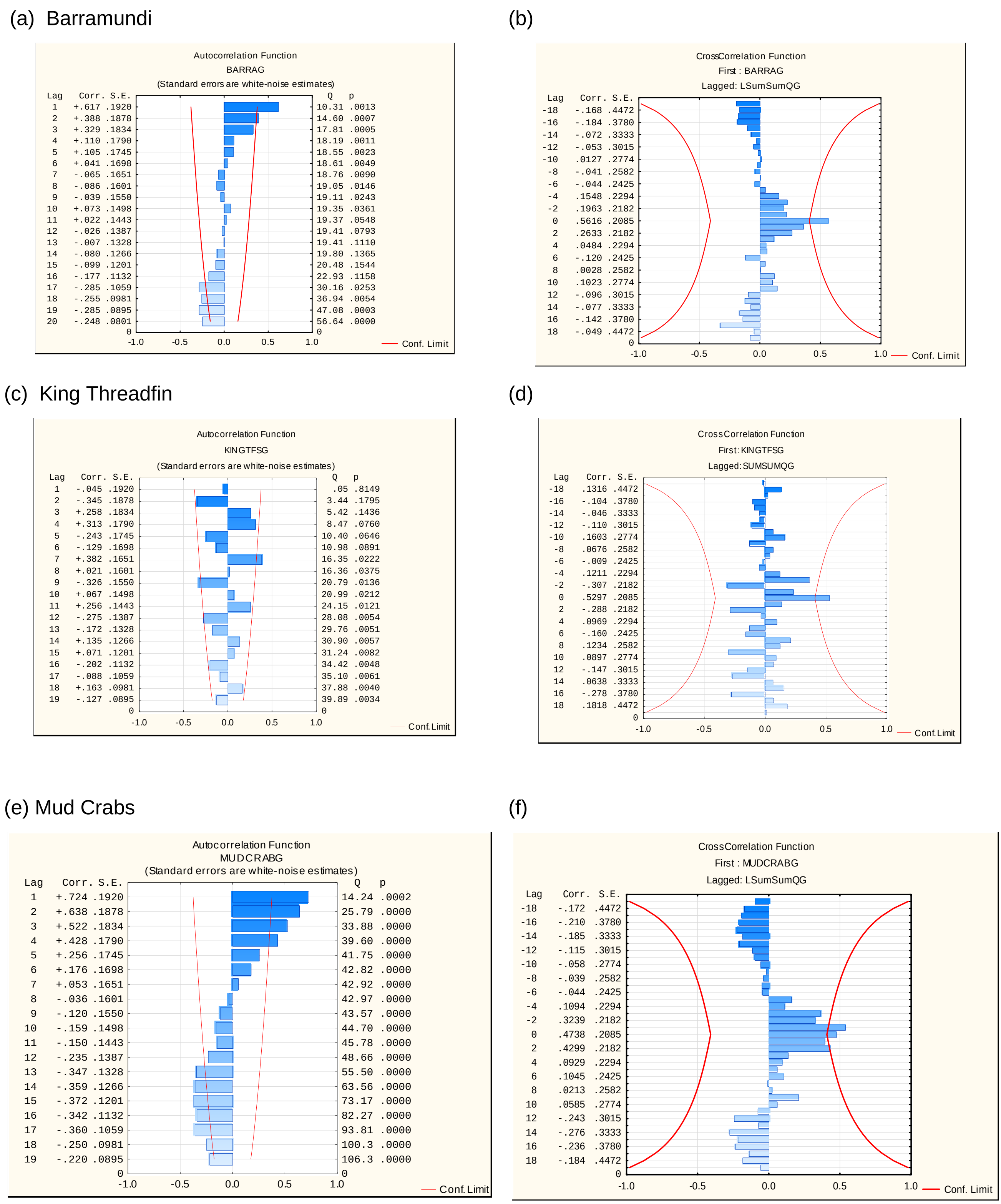


Figure 3.19 a-f. Time series analyses of CPUE data ($\log_{10}$ weight/boat days) of selected species in the Gilbert River (1989-2011). Data are species caught, catch (t) and effort (fishing days using pots or nets). Auto correlation and cross correlations with wet year flow (ML) are Barramundi (a&b), King Threadfin (c & d) and Mud Crabs (e&f), respectivly.

13.1.2 RISK ASSESSMENT APPROACH AND MODELS PREDICTING CATCH FROM EFFORT AND FLOW

The same risk assessment framework and statistical modelling methods used for Banana Prawns were used for Barramundi in both rivers. Catch and effort data were paired to a wet year (October to September for the wet years 1988/1989 to 2009/2010, n=22). Normal probability plots, and Kolomorov-Smirnov and Shapiro-Wilks' W test (Zar 1974), all suggest that catch data are essentially normally distributed. Hence, these data were not transformed.

13.2 Results

13.2.1 BARRAMUNDI, KING THREADFIN AND MUD CRAB

The multiple regressions equations used to predict Barramundi catch (t) in the Gilbert and Flinders rivers from fishing effort (Boat days) and Wet Year flow (ML, Q; 1989-2010) are summarised in Table 3.6 and Table 3.7, respectively. Effort and flow had similar influences on observed catch in the Gilbert River but, in contrast, effort explained 6 times more variation in catch than flow in the Flinders River.

Table 3.6 Summary of the multiple regression equation used to predict commercial Barramundi catch (log_{10}Weight, t) from fishing effort (log_{10}E, boat days) and wet year flow (log_{10}Q, ML) in the Gilbert River (1989-2010).

R= 0.780, adjusted R^2 = 56.7%, n= 22, P< 0.001, SE regression = 12.33.

Variable	Beta	SE Beta	B	SE B	P
Intercept			-190.95	52.89	=0.002
Q (Flow)	0.540	0.144	28.34	7.813	<0.001
E (Effort)	0.564	0.144	0.047	0.013	<0.001

Table 3.7 Summary of the multiple regression equations used to predict commercial Barramundi catch (log_{10}Weight, t) from fishing effort (log_{10}E, boat days) and wet year flow (log_{10}Q, ML) in the Flinders River (1989-2010).

R= 0.947, adjusted R^2 = 90%, n= 22, P< 0.001, SE regression = 0.10

Variable	Beta	SE Beta	B	SE B	P
Intercept			-181.6	35.67	=0.006
Q (Flow)	0.172	0.140	13.34	5.75	=0.032
E (Effort)	0.907	0.141	0.074	0.006	<0.001

Table 3.8 Summary of the multiple regression equation used to predict commercial King Threadfin catch (log_{10}Weight, t) from fishing effort (log_{10}E, boat days) and wet year flow (log_{10}Q, ML) in the Gilbert River (1989-2010).

R= 0.874, adjusted R^2 = 73.4%, n= 22, P< 0.001, SE regression = 5.64

Variable	Beta	SE Beta	B	SE B	P
Intercept			-62.47	25.34	=0.023
Q (Flow)	0.302	0.116	9.52	3.67	=0.018
E (Effort)	0.879	0.116	0.070	0.009	<0.001

The multiple regression equation used to predict King Threadfin catch in the Gilbert from fishing effort and wet year flow (1989-2010) is summarised in Table 3.8. Effort explained about three times more variation in catch than flow.

Table 3.9 Summary of the multiple regression equation between commercial Mud Crab catch (log_{10}Weight, t) from fishing effort (log_{10}E, Pot days) and wet year flow (log_{10}Q, ML) in the Gilbert River (1989-2010).

R= 0.868, adjusted R^2 = 73%, n= 22, P< 0.001, SE regression = 7.09

Variable	Beta	SE Beta	B	SE B	P
Intercept			-42.37	10.69	<0.001
Q (Flow)	0.371	0.120	12.60	3.80	<0.001
E (Effort)	0.737	0.120	0.044	0.007	<0.01

The multiple regression equation used to predict Mud Crab catch in the Gilbert from fishing effort and wet year flow (1989-2010) is summarised in

Table 3.9. Effort explained twice the variation in catch than flow.Effort explained most variation in Mud Crab catch in the Flinders (R^2 = 97%, n=22, P<0.001), and flow was not significant on its own or in combination with effort. Similarly, King Threadfin catch in the Flinders showed no correlation with flow and a significant, albeit mediocre, correlation (R^2=33%, n=22, P<0.01) with effort (boat days), perhaps indicating that fishers were primarily targeting Barramundi. The single regression relationships between flow and effort for Barramundi in the Gilbert and Flinders rivers are illustrated in Figure 3.20a-d, respectively.

The partial residuals plots of the regression equations predicting Barramundi catch from fishing effort and wet year flows in the Gilbert and Flinders are illustrated in Figure 3.21a-d, respectively. These plots show the influence of each variable in the joint model when the influence of the other variable is held constant at its mean value. The effect of flow on catch in each river is enhanced when the effect of effort is factored out.

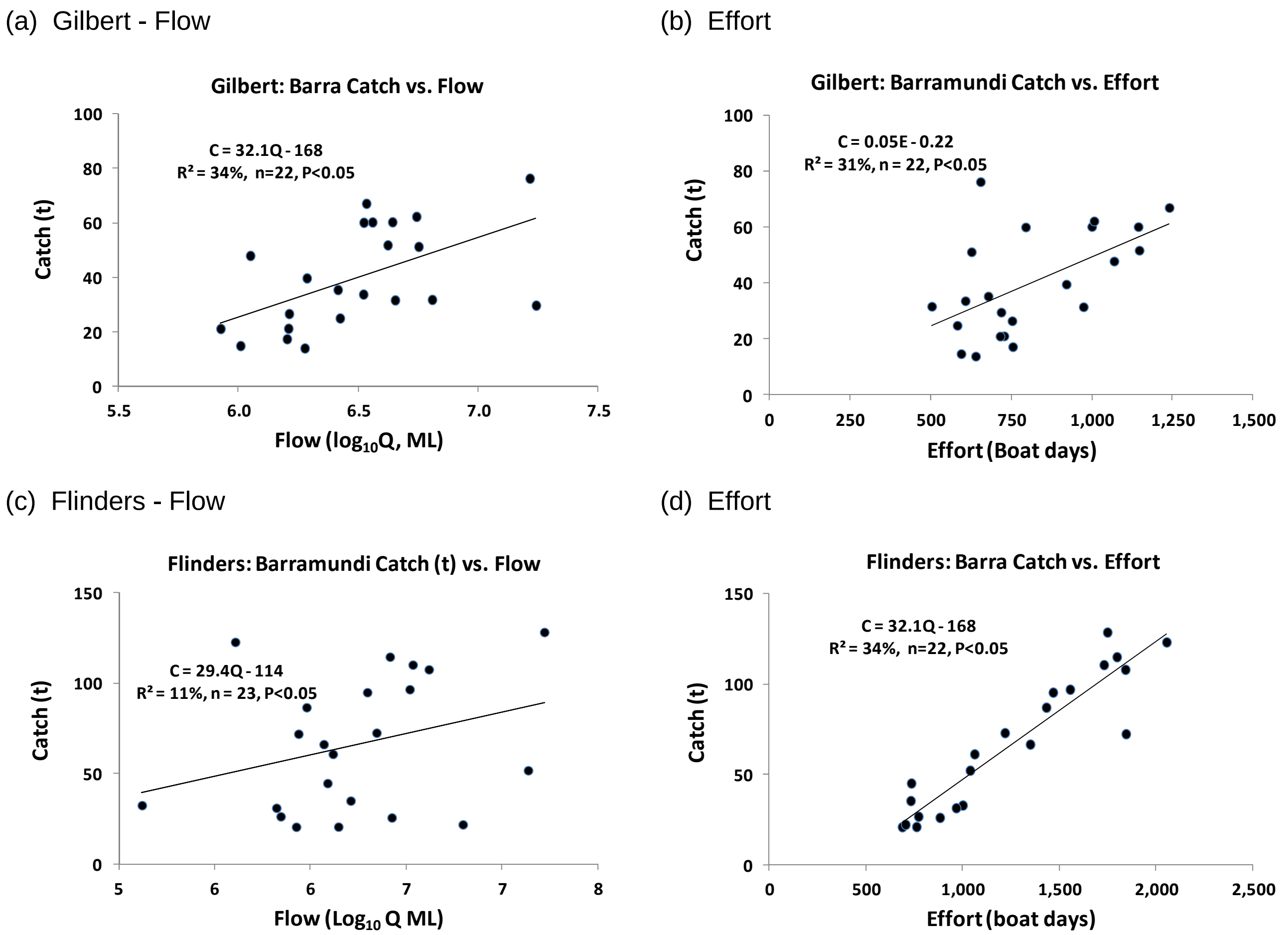
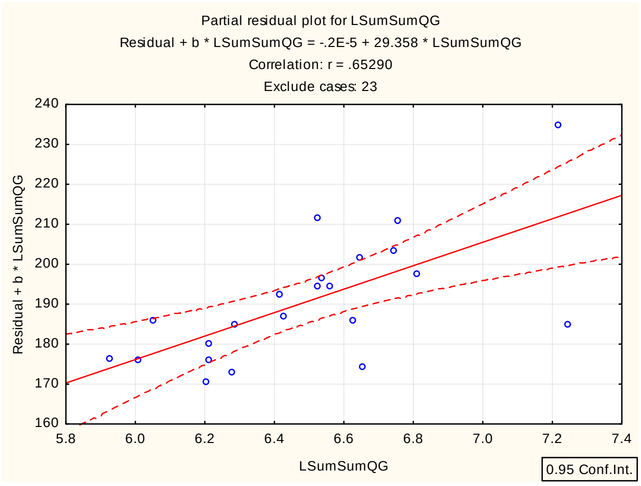
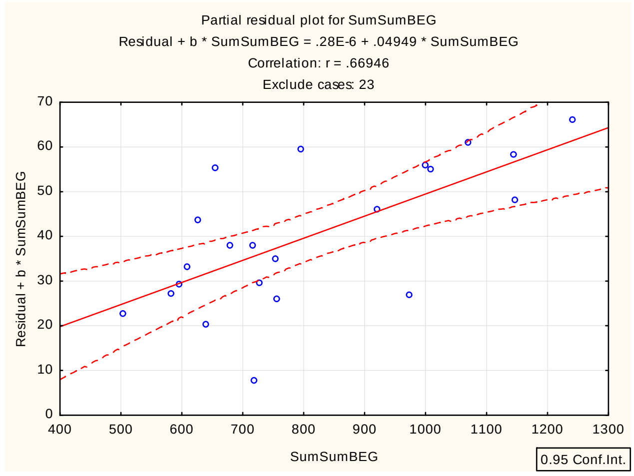


Figure 3.20 a-d. Linear regressions predicting Barramundi Catch (C, t) from Wet Year flow volume (Q, ML) and fishing effort (E, Boat days) for the Flinders River (a & b, respectively) and Gilbert River (c & d, respectively). Flow data were transformed to $\log_{10}$ and catch data untransformed (see text).

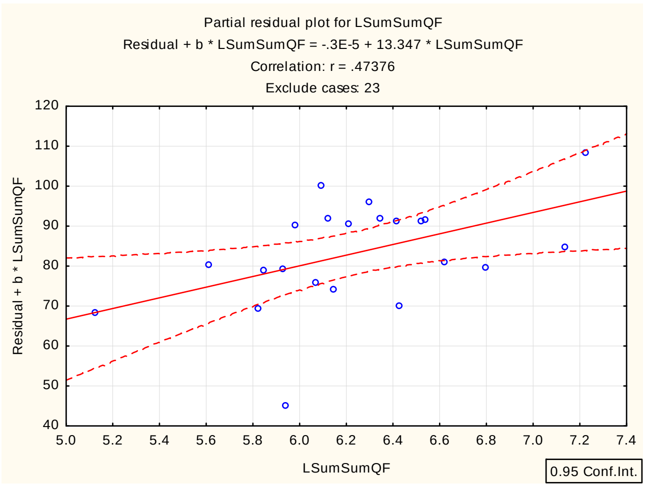
(a) Gilbert - Flow



(b) Effort



(c) Flinders – Flow



(d) Effort-

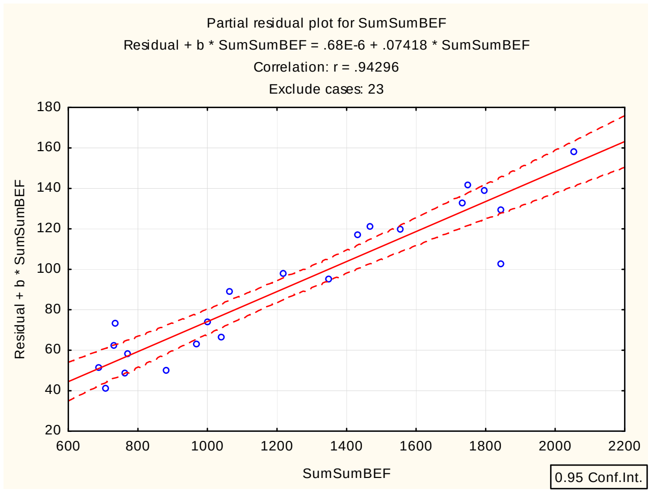
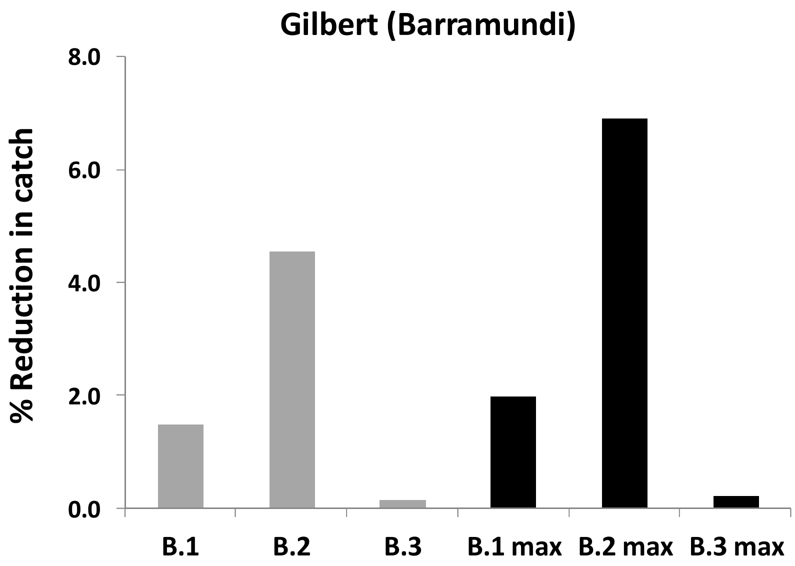


Figure 3.21 a-d. Partial residuals plots of the multiple regression equation predicting Barramundi catch (t) from flow (log_{10}Q ML) and effort (boat days) for the Gilbert (a&b) and Flinders (c&d) rivers (1989 – 2010). Results show the influence of each variable on catch with the other variable held constant at its mean value.

13.2.2 PREDICTED REDUCTION IN BARRAMUNDI CATCH FOR FGARA DEVELOPMENT SCENARIOS

The equations derived above were used to predict the reduction (%) in the mean (1989-2010) Barramundi catch in the Flinders and Gilbert rivers for each FGARA development scenario (i.e. water harvest from annual yield to maximum storage capacity, see Table 3.2). The predicted reduction in catch in the Gilbert (Figure 3.22a) is 5% or less for all B scenarios using annual water yields. Where we assume that maximum storage capacity is used instead of annual yields, the predicted reduction in catch is 7% or less. The predicted reduction in catch in the Flinders (Figure 3.22b) is negligible at 0.2% or less for all B scenarios. Where we assume that maximum storage is used the predicted reduction in catch is 3% or less.

(a)



(b)

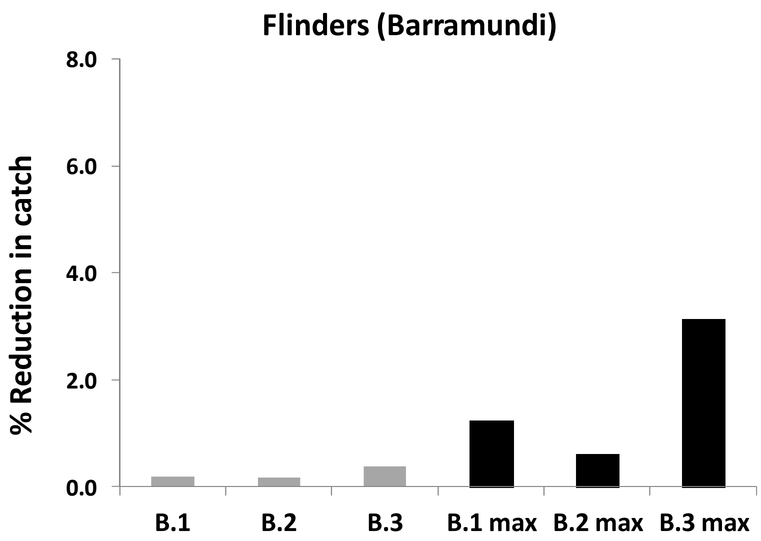


Figure 3.22 a&b. Predicted reduction (%) in mean (1989-2010) Barramundi catch in the (a) Gilbert and (b) Flinders rivers, for each FGARA development scenario (i.e. from annual expected water harvests (grey) to the maximum storage capacity (black).

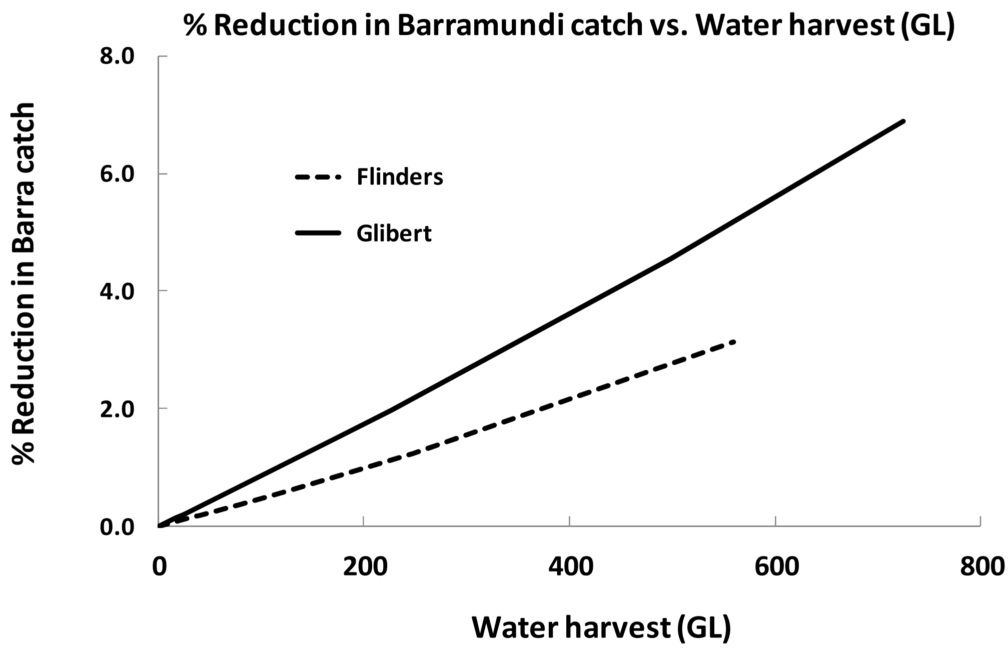
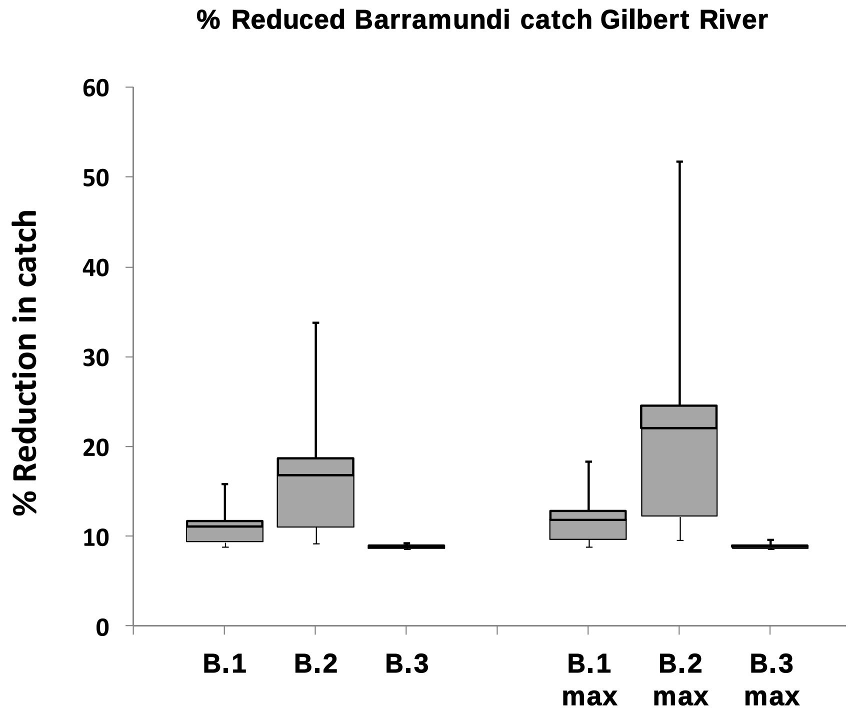


Figure 3.23 Predicted reduction (%) in mean (1989-2010) Barramundi catch (t) from reductions in EOS flow (GL) due to expected water needs of each FGARA B development scenarios (ranging from predicted annual water yield to maximum capacity of storages).

The simplistic relationship between water harvest and predicted reduction (%) in mean (1989-2010) Barramundi catch in the Gilbert and Flinders rivers is plotted in Figure 3.23. However, as with the Banana Prawn catch-flow-effort model these results do not account for model error (including Source model errors) or intrinsic variability in effort and flow. Nor can they be extrapolated to sustained periods of low-flow conditions as experienced in a prolonged drought. The results of the uncertainty analysis for both rivers are illustrated in Figure 3.24 and provide a range of estimates for reductions in catch for each FGARA B scenario (after 10,000 Monte Carlo simulations). For the Gilbert B.2 scenario that had the greatest water harvest and used expected annual yields, 50% of simulations ranged between a 3 and 12% (mean 10%) reduction in catch. Where we assumed maximum storage capacity was used the range was a 4 to 19% (mean 16%) reduction in catch. For the Flinders B.1 and B.2 scenarios that used expected annual yields for in-stream storage, 50% of simulation results were <2% reduced catch with negligible range. Where we assumed maximum storage capacity for these two scenarios, results were just above a 2% reduction in catch, also with negligible range. Where we assume that the maximum entitlement is used for off-stream irrigation/storage in scenario B.3, 50% of simulations ranged between a 2 and 4% (mean of 3%) reduction in catch.

(a)



(a)

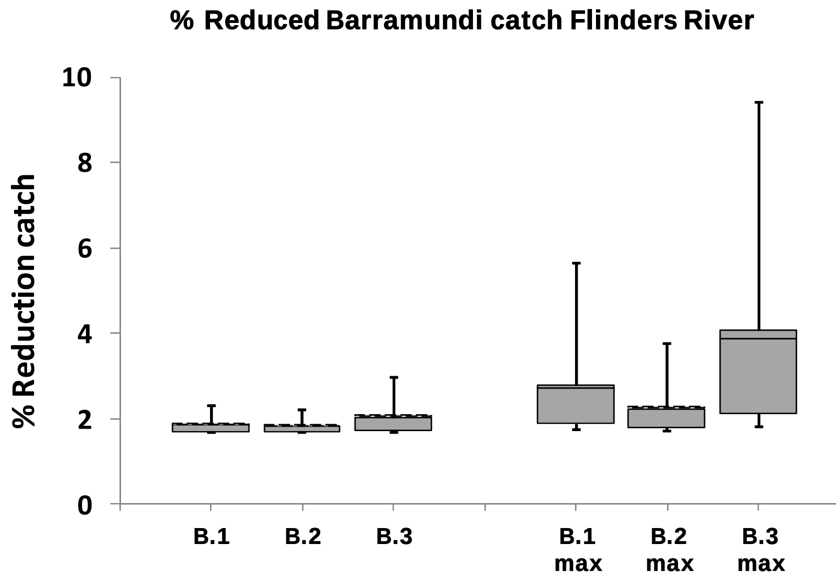


Figure 3.24 a&b. Predicted percentage (%) reduction in mean (1989-2010) Barramundi catch in the (a) Gilbert and (b) Flinders for each FGARA development scenario (see text) when accounting for uncertainty in models. Grey boxes encompass values for 50% of all simulations (25% to 75%), the dark horizontal lines are mean values and the vertical lines represent the 5% to 95% range.

13.3 Year Class Strength (YCS) - Barramundi

13.3.1 INTRODUCTION

The Barramundi fishery in the south-eastern GoC and its corresponding catch are underpinned by its population dynamics, which is highly dependent on river flows. Barramundi recruitment is therefore likely to be highly influenced by changes in river flow that are either natural and/or human imposed. Hence, modelling the relative recruitment of Barramundi may provide a means of assessing the relative impact of differing water resource management scenarios and has been used elsewhere in Queensland (DSITIA 2013). Barramundi recruitment can be retrospectively estimated from the age-structure of those caught in the fishery. Staunton-Smith et al. (2004) provides detailed methods, assumptions and limitations of estimating relative Year Class Strength (YCS), which is used here to assess the impacts of modelled Gulf WRP water use scenarios on Barramundi populations in the Flinders and Gilbert rivers. For both rivers the Gulf WRP Pre-development and Current scenarios (Table 3.10) are equivalent to FGARA's Natural and current (Scenario A, Table 3.2) scenarios. For the Flinders, the Gulf WRP Scenario 2 (Table 3.10) is equivalent to FGARAs Flinders Scenario B.3 max (560GL, Table 3.2) extraction for off-stream/on-farm irrigation/storage. These new entitlements are in addition to the 80GL of water released in 2013 and to the pre-2013 entitlement of 25GL (see Part 1, Section 2.1.2). For the Gilbert, the Gulf WRP Scenario 1 (Table 3.10) is equivalent to FGARAs Gilbert B.2 Scenario (725GL, Table 3.2, Dagworth and Green Hills dams). The Gulf WRP Scenario 2 for the Gilbert (Table 3.10) is Scenario 1 plus non-irrigation demands (e.g. town water supplies and mining).

Table 3.10 Summary statistics for modelled End-of-System wet year flow (Oct. to Sept., ML) characterising water use scenarios of the current Gulf WRP, based on the most downstream gauging station in each catchment for the time series 1890 to 2010. Data supplied to DSITIA by DNRM.

Catchment	Gilbert @ Miranda Downs 9170094				Flinders @ Walkers Bend 915003A		
Gulf WRP	Pre-development	Current	Scenario 1 (725 GL additional harvest)	Scenario 2 (~725 GL additional harvest)	Pre-development	Current	Scenario 2 (560 GL additional harvest)
FGARA	Natural	Scenario A	B.2 max	B.2 max	Natural	Scenario A	B.3 max
Variable							
Mean (ML)	3,775,190	3,734,280	3,180,216	3,098,840	2,695,151	2,617,981	2,363,148
Std Dev	4,257,120	4,275,641	4,229,295	4,214,487	3,776,012	3,755,541	3,681,594
Skewness	4.00	4.05	4.24	4.28	3.00	3.00	3.13
Kurtosis	24.4	24.8	26.4	26.8	11.9	12.1	13.0
Median (ML)	2,646,121	2,603,484	1,931,275	1,824,432	1,409,247	1,305,533	988,175
Minimum (ML)	82,859	54,590	39,877	39,307	<1	<1	<1
Maximum (ML)	35,118,344	35,348,883	34,919,218	34,817,320	25,094,529	24,984,677	24,717,558
Count	121	121	121	121	121	121	121
10th Percentile	509,988	468,451	237,258	213,947	71,236	52,667	5,252
90th Percentile	7,880,363	7,847,632	7,287,161	7,180,847	7,162,727	7,049,380	6,677,110

13.3.2 METHODS - RELATIONSHIP BETWEEN BARRAMUNDI YCS AND FLOW

Flinders River

Age-structure data of commercially caught Barramundi were available from data collected by Halliday et al. (2012). The data enabled a local equation to be developed for the Flinders River, using local estimates of both relative YCS and modelled Gulf WRP flow data (Table 3.10). Barramundi YCS between 2002 and 2008 (inclusive) was screened against modelled current use flow at gauging station 915003A, (i.e. the most downstream gauging station - Walkers Bend representing End-of-System flow) to identify the month(s) of flow that explained the observed variation in relative YCS. The model with the highest adjusted R^2 (from the General Linear Model of average YCS against flow) and that was plausible in terms of the known biology of Barramundi was selected for assessment of water-use scenarios in the Flinders River (Table 3.11).

Table 3.11 Model parameters and equation for use in the assessment of the risk of water development scenarios to Barramundi populations in the Flinders and Gilbert rivers.

Catchment	Flow variable (Total, ML)	YCS Equation	$\%R^2$
Flinders	[Feb + Mar + Apr]	Average YCS = 0.719*Log10[Feb + Mar + Apr Flow] – 4.088	80.3
Gilbert	[Apr]	Average YCS = 0.676*Log10[Apr Flow] – 3.03	43.3

Gilbert River

There is no readily available calculation of Barramundi YCS in the Gilbert River. Therefore, any assessment of water use scenarios on Barramundi YCS in the Gilbert River requires some assumptions. Examination of monthly flows indicated that the Gilbert River had longer periods of seasonal flows and much shorter cease to flow periods than the Flinders River (i.e. maximum no-flow spell duration of 149 days compared to 662 days under the current use scenarios). Barramundi YCS data were readily available from the Mitchell River, which is to the north of the Gilbert River and whose monthly flow pattern was more similar to the Gilbert than that of the Flinders (Figure 3.25). Hence, in an attempt to assess water-use scenarios in the Gilbert River, we assumed that the relationship between Barramundi YCS and flow were similar for the Mitchell and Gilbert Rivers. This is an assumption that requires validation and may be possible through the analysis of the age-structure data for barramundi caught in the Gilbert River and which resides with Fisheries Queensland. However, such analysis is beyond the capacity of the current project. Age-structure data for Barramundi caught in the Mitchell River collected by Halliday et al. (2012) were reanalysed for the current assessment. Barramundi YCS between 1999 and 2007 (inclusive) were screened against: (i) gauged flow at Gamboola (919001A) in the Mitchell River; and (ii) modelled WRP current use flow at Miranda Downs in the Gilbert River to order to identify whether or not month(s) of flow that explained the greatest amount of variation in YCS were the same across the two river catchments. The model with high adjusted R^2 (from the General Linear Model of Average YCS against flow), plausibility in terms of known Barramundi biology, and consistency across catchments was selected for assessment of water use scenarios in the Gilbert River (Table 3.10).

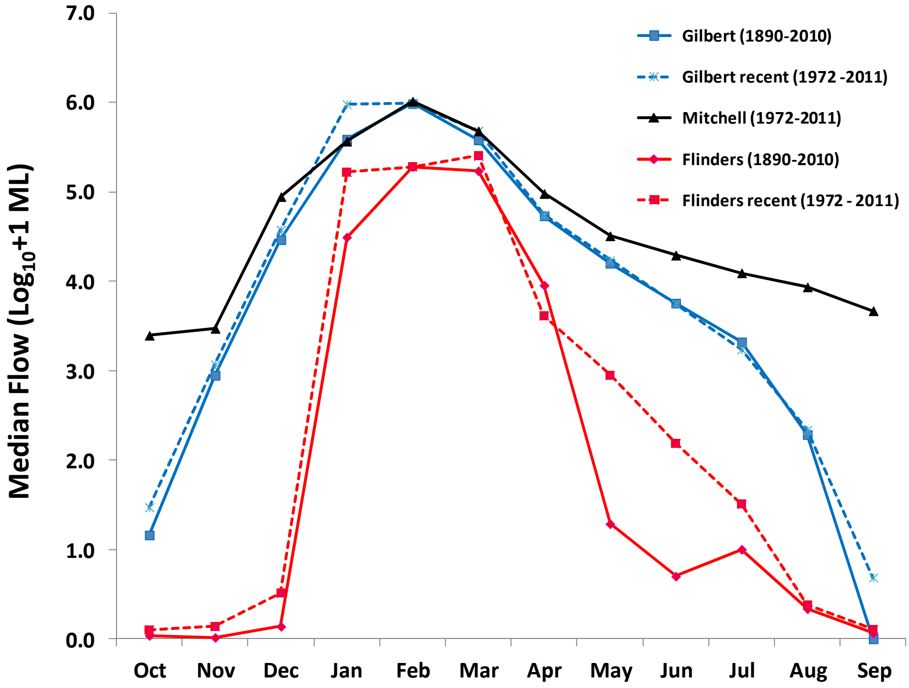


Figure 3.25 Monthly pattern of median flow (Log_{10} + 1) within a 'wet year' (Oct. to Sept.) for the Gilbert, Mitchell and Flinders rivers. Gilbert and Flinders flow data are for the modelled Gulf WRP current-use scenario, whilst the Mitchell is gauged flow at Gamboola (919001A).

Barramundi YCS was fitted against the yearly time series of End-of-System flow for the Flinders and Gilbert catchments supplied and fitted by the Water Planning Ecology group of DSITIA. Model outputs were a yearly time series of relative YCS for each water use scenarios identified in Table 3.10.

Thresholds of Concern

We adopted the same Threshold of Concern (ToC) as defined in the Wet Tropics WRP Environmental Assessment, Appendix B (DSITIA 2013). That is, that YCS is fundamentally related to the productivity of Barramundi populations. The median value of Barramundi YCS for a modelled Pre-development (PD) flow sequence was assumed as the threshold above which recruitment of Barramundi was sufficient to support the long-term viability of the population. We also applied the same risk categories as those used in the Wet Tropics WRP (Table 3.12). That is, a high risk category is equivalent to potential local population failure and was defined as a period where the number of consecutive years without sufficient recruitment exceeded 11 years ("the longevity of the species"). A moderate risk profile was defined as the period where the number of consecutive years without sufficient recruitment was between 5 years ("the average age of protandry" – minimum age of sex inversion from males to female, based on Moore 1979) and 11 years. Periods less than 5 consecutive years without sufficient recruitment were deemed as having low risk to potential local population failure.

Table 3.12 Categories of risk to potential local population failure of Barramundi based on the sequence of projected relative Year Class Strength.

Risk category	Consecutive years YCS < median value for modelled Pre-development flow
Low	< 5
Moderate	5 to 11
High	>11

13.3.3 RESULTS

Flinders River

The average relative YCS of Barramundi in the Flinders River over the simulation period (n=121 years) was reduced by 23% and 71% for the current and full-development scenarios respectively, as compared to the pre-development scenario. When considered in relation to the Thresholds of Concern (ToC), changes in YCS as a consequence of changes in EOS flow magnitude and timing resulted in a 1% and 15% increase in the number of years in the moderate risk category under the current and full-development scenarios respectively (Figure 3.26). There were no years in the high risk category under the pre-development or current-use scenarios, but there was a 3% increase in the number of years in the high risk category (i.e. with the potential for local population failure).

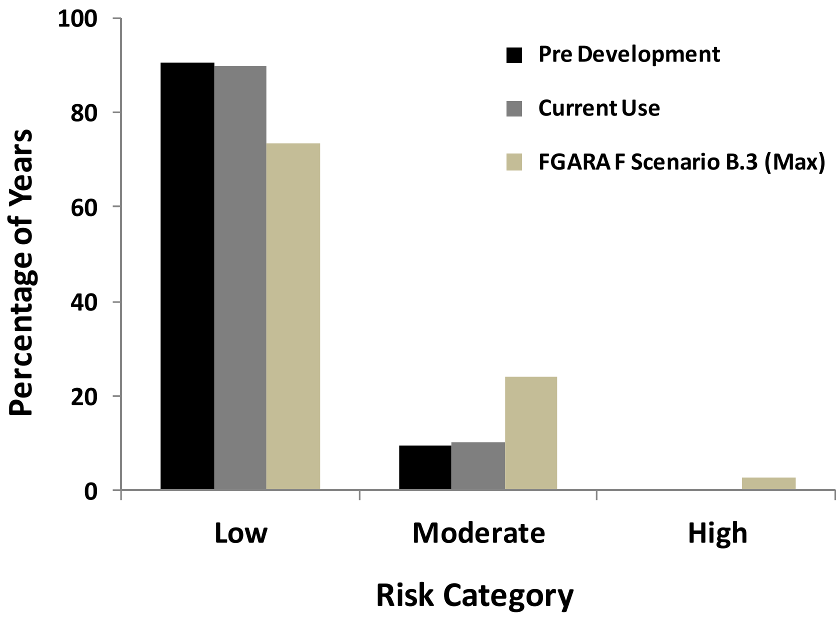


Figure 3.26 Risk profile for Barramundi in the Flinders River as a percentage of years in the simulation period (n=121) in each risk category for Thresholds of Concern.

Gilbert River

The average relative YCS of Barramundi in the Gilbert River over the simulation period (n=121) was reduced by 16% for the current-use scenario, by 172% for the full-development Scenario 1 and by 175% for full-development Scenarios 2, as compared to the average YCS of the pre-development scenario. When considered in relation to the Thresholds of Concern, the magnitude and timing of these decreases resulted in a 3% increase in the number of years in the moderate risk category for the current-use scenario and a 21% increase in the number of years in the moderate risk category for both full-development scenarios (Figure 3.27). There were no years in the high risk category under the current-use scenario, but there was a 12% increase in the number of years in the high risk category (i.e. local population failure) for both full-development scenarios.

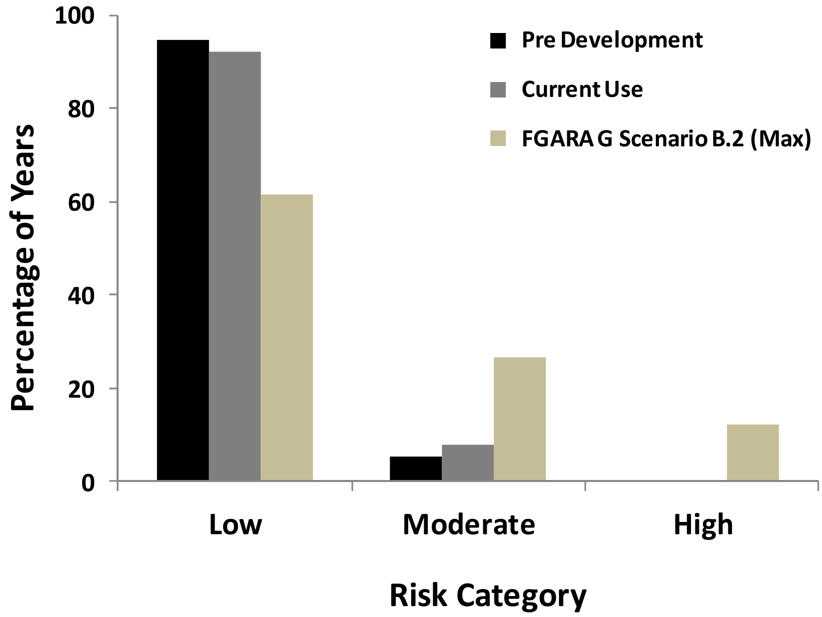


Figure 3.27 Risk profile for Barramundi in the Gilbert River as a percentage of years in the simulation period (n=121) in each risk category for Thresholds of Concern.

13.3.4 DISCUSSION OF YCS RESULTS

Water flowing to the ocean is not wasted, but rather supports productivity that is currently exploited by several downstream stakeholder groups. For example, Barramundi is a flow-dependent commodity that is valued commercially, recreationally and by Indigenous people. Upstream water-use will need to be traded-off against downstream productivity.

Sufficient recruitment is vital to: (i) maintain a viable Barramundi population in a river system; and (ii) produce sufficient biomass of Barramundi that can be used sustainably. The Thresholds of Concern used in the above analysis are relevant to the existence of a viable Barramundi population. The water-use scenarios proposed for full-development in the Gilbert River had the greatest increase in the number of years in the high risk category (from 0 to 12%). Water-use scenarios proposed for full-development in the Flinders River had a much smaller increase in the number of years in the high risk category (from 0 to 3%). The high risk category Threshold of Concern relates to the risk of “potential for local population failure”. It is noteworthy that in the Gilbert and Flinders rivers, the proposed water-use scenarios create the possibility of local population failure, but this was never a risk in either system under current or pre-development water-use scenarios. Barramundi populations have evolved to deal with the flood drought cycles of northern Australia, and thus far have survived several long periods of drought and consequential low flows. Whilst Barramundi populations are resilient to variability in river flow, it remains to be seen as to whether or not Barramundi populations could recover (and at what rate) from naturally occurring years of low-flow that are extended as a consequence of the proposed full-development water use scenarios. Barramundi populations do exist in parts of northern Australia where there is minimal river flow into the sea (e.g. parts north Western Australia), but these populations are small in size and support limited fisheries.

A sustainable fishery depends on exploiting the biomass of a species that is in excess of its needs to replenish its population. Much of the fisheries management of Barramundi (and the GoC Inshore Finfish Fishery) is aimed at ensuring that a sufficient part of the population is unharvested to enable the population to replenish itself. It is highly likely that the moderate risk category (i.e. the period where the number of consecutive years without sufficient recruitment is between 5 years and 11 years) is more representative of likely impacts of upstream water use on Barramundi fisheries in the Gilbert and Flinders rivers. Because of the relationship between river flow and YCS, increases in the frequency of the moderate risk category will be associated with a decline in Barramundi production—and thereby increase the frequency of reduced Barramundi catches by commercial, recreational and Indigenous fishers. The number of years in the time series of YCS classified as at moderate risk is trebled in the Gilbert River and doubled in the Flinders River (Figure 3.27 and Figure 3.26, respectively) compared to the current water-use scenario. This will significantly increase the number of years when Barramundi catches are reduced and therefore the fishery (especially the commercial sector) may have compromised profitability and viability (in both the short and long term) because the proposed full-development water use scenarios have impeded the natural flow pulsed recruitment that replenishes Barramundi populations.

The current YCS analysis could be improved through: (i) dedicated analysis of age-structure data held by Fisheries Queensland to extend YCS estimates in the Flinders River to cover recent flood years; and (ii) developing a local equation for YCS and flow for the Gilbert River to validate whether or not the Gilbert River YCS behaves in a similar manner to that of the Mitchell River YCS, which is assumed here. A local model linking flow to Barramundi YCS could more thoroughly explore which aspects of the flow regime are important for Barramundi recruitment. In the model assumed for the Gilbert River, only April flows are considered, as a consequence transferability of the models with high adjusted R^2 from the Mitchell to the Gilbert River. It is likely that flow in April is a surrogate for wet season flows of longer duration or greater flood plain inundation, or extended connection between the estuary and upstream freshwater reaches thereby enhancing young-of-the year survival. Barramundi is an opportunistic species that exploits different conditions provided by different flow regimes in different rivers. This is a topic that requires further research and where local understanding and equations would significantly benefit the assessment of water-use scenarios. It should be noted that Fisheries Queensland collect age-structure data on Barramundi to monitor genetically defined regional stocks, and that this data may have insufficient temporal and spatial resolution to address YCS estimates for individual rivers.

Future monitoring of Barramundi age-structure in the Gilbert and Flinders rivers is recommended to validate that the Thresholds of Concern used in the current analysis appropriately represent the risk to Barramundi from water-use scenarios, with respect to maintaining the viability of these populations as well as the fisheries dependent upon them. Additional monitoring on Barramundi catch and movement, and profitability of Barramundi fisheries, would assist in understanding projected and actual impacts of upstream water use on downstream stakeholders.

13.4 Discussion and Recommendations

Knowledge of the relationship between freshwater flow and fishery species in tropical and subtropical estuaries of Queensland, and its implications for managing flow allocations, have been extensively investigated by Robins et al. (2005), Robins et al. (2007b), Robins and Qifeng (2007) and Halliday and Robins (2007), and by Halliday et al. (2012) for estuarine finfish fisheries in the GoC. Of direct relevance to our fisheries assessment in the GoC. Robins et al. (2007a) developed a conceptual framework to better understand the freshwater needs of estuaries for sustainable fisheries production in order to assess the impacts of water regulation. Their conceptual framework identified aspects of the freshwater flow regime that are potentially important for estuarine species, and was used to propose hypotheses of key biological processes that are influenced by, or dependent on, freshwater flow. They then refined these hypotheses using a combination of life history information and analysis of catch and freshwater flow data, a process similarly adopted in this project for other species. They concluded that whilst numerous correlative studies demonstrate that catch of some fishery species are strongly linked to freshwater flow, the causal mechanisms underlying these relationships need to be better understood in order to develop mitigation strategies to minimise the impacts of altered freshwater flow regimes.

The correlation between Barramundi catch and total wet year flow volume found in this study for the Flinders and Gilbert rivers suffer the same constraints in that the causal mechanisms that underpin it are not well understood, if at all. The degree to which catch-adjusted-for-effort reflects population level responses to flow is unknown. For example, Scenario B.2 in the Gilbert catchment had the greatest water harvest of those that used expected annual yields (498 GL, or 14% of the mean and 19% of the median wet year flow), and simulated Barramundi catches declined by up to 12%. In contrast, Scenario B.3 max in the Flinders catchment had the greatest water harvest being the maximum entitlement proposed for off-stream/on-farm storage for irrigation (560 GL, or 21% of the mean and 41% of the median wet year flow), yet simulated catches only declined by up to 4%. Whilst the latter results suggest that Barramundi catch in the Flinders may be relatively resilient to decreased flow, alternatively they may be indicative of a population with poor recruitment due to heavy fishing (e.g. fishing effort doubled since 2000). Effort and flow had similar influences on observed catch in the Gilbert River and, in contrast, effort explained six times more variation in catch than flow in the Flinders River. Furthermore, Bayliss et al. (2008) found that recreational and commercial catches of Barramundi in the Daly River of the NT were positively correlated with flow at 0, 2 and 3y time lags. They suggested that the positive correlation between instantaneous catch and flow may encompass a large component of increased catchability due to increased movements, in addition to increased recruitment and/or survival effects. They speculated that this cohort may comprise mostly upstream males migrating to the river mouth to spawn. In contrast, they argued that their positive correlations at 2 and 3 year time lags would mostly reflect enhanced recruitment and/or survival of the cohort that has reached the size limit to enter the fishery (55cm total length for NT fisheries). Time series analysis detected no positive correlation between Barramundi CPUE and previous wet year flows in the Flinders and Gilbert data. This highlights that fisheries-independent population level data, such as the Year Class Strength (YCS) analyses reported above, are needed to support our simulation results of the impact of water extraction scenarios on catch success. Such population structure data are not confounded by movements and, hence, may better identify critical stages in the life-history-flow cycle for management of this species.

The risk to Barramundi population recruitment from FGARA development scenarios as indexed by their YCS was quantified also, given that DAFF data were available and it had previously been used in the Environmental Assessment of the Wet Tropics Water Resource Plan. Local relationships (equations) between Barramundi YCS and flow were developed for the Flinders and Gilbert rivers and then used to predict changes in YCS under pre-development (Natural), current (Scenario A) and proposed full-use water development scenarios (i.e. Flinders Scenario B.3 max and Gilbert Scenario B.2 max). Barramundi populations were considered at high risk of local population failure if modelled YCS was less than the median YCS for the modelled pre-development flow sequence for greater than 11 consecutive years. If modelled YCS was less than the median YCS for 5 to 11 consecutive years then populations were considered at medium risk. Results for YCS in both rivers show that no years were classified as high risk for

Barramundi populations under pre-development and current water-use scenarios. However, in contrast, under full water-use scenarios the high risk category increased from 0 to 3% in the Flinders River, and from 0 to 12% in the Gilbert River. Whilst Barramundi have evolved to deal with the flood-drought cycles of northern Australia, the proposed full development water-use scenarios may artificially extend natural periods of low-flow to the estuary reducing favourable conditions for recruitment and, hence, population replenishment. Whilst the results are similar to the percentage reductions in catch using catch-flow models summarised above, under full water-use development scenarios the moderate risk category increased from 10 to 24% in the Flinders River, and from 5 to 26% in the Gilbert River, suggesting that the long-term sustainability of the Barramundi fishery as a whole could be at significant risk unless mitigation strategies are adopted, such as those recommended in Part 4 of this report.

Our correlation model between Barramundi catch and total wet year flow may also not account for critical thresholds in the frequency and duration of floodplain floods, which are essential to the life-history strategy of this species (Part 2 Section 8.1). For example, Sawynock and Platten (2011) found that increased duration between flood events had negative effects on Barramundi recruitment. The Source model outputs (Petheram et al. 2013a,b) for the Flinders and Gilbert since 1889 include daily estimates of the area of floodplain floods using a spatially explicit hydrological model, and these flow-related variables should be examined in more detail for relationships with commercial catch. For example, Barramundi CPUE in the Flinders River had a stronger positive correlation with minimum wet year flow than with total wet Year flow, and this analysis should be pursued as it may indicate a threshold flow value for water management purposes. The correlation between the minimum flow in any wet year and total wet year flow is essentially a concave nonlinear relationship for both river systems.

Mud Crabs in the Gilbert had positive a correlation with flow and, similarly for King Threadfin. Both species had no correlation with flow in the Flinders and these are inconsistent results, possibly reflecting nuances in the catch-effort data. However, our results for Mud Crabs in the Gilbert support the study of Meynecke et al. (2011), who examined variations in commercial catch across northern Australia and found that rainfall-river flow explained 30-70% of the variability in catch. Halliday et al. (2008) found also that freshwater flow affects the YCS of King Threadfin in some tropical estuaries, supporting the flow-catch relationship for this species in the Gilbert. Barramundi and King Threadfin dominate the catch in both rivers and, given their strong relationship with flow, would therefore make ideal indicator species for monitoring potential impacts from sustained water extractions for future agricultural developments such as FGARA.

13.4.1 RECOMMENDATIONS

i. Population level data may be required to assess potential impacts of flow extraction on fisheries production given the possible confounding of movements with *in situ* recruitment and survival in the catch and effort data. We recommend that YCS studies be continued in the Flinders and initiated in the Gilbert for Barramundi and King Threadfin, as they dominate the catch and would make ideal indicator species for monitoring potential impacts from future agricultural developments such as those proposed by FGARA.

ii. For all flow-dependent species in the GoC fisheries, the influence of the extent, frequency and duration of floodplain floods in relation to catch needs to be examined in greater detail. There may be possible low-flow relationships that water managers can use as a Threshold of Concern (ToC, DSITIA 2013). Total wet year flow volume may not explain all critical population processes of Barramundi and, consequently, their future abundance and catch.

iii. The full economic impact of sustained reductions in Barramundi catch is not addressed here given time constraints, but is a critical part of any assessment of water tradeoffs between agricultural and fisheries production. The economic and non-market impacts of water extraction on Indigenous and recreational catch were not part of this assessment, and should be included in future quantitative assessments.

iv. Further consultation and engagement with the GoC fisheries sector, including recreational and Indigenous fisheries, are required to communicate our assessments accurately with respect to the true uncertainties in the risk models. The risk models and simulations are simply “what if” future scenarios, not accurate real world predictions. They are also underpinned by untested assumptions in the predictive catch-effort-flow models used here.

v. The Indigenous cultural values of coastal and marine fisheries in the GoC are unknown (Barber 2013; pers. comm.), although Jackson et al. (2011) has documented important Indigenous socio-economic values in relation to river flow. Consultation and engagement of coastal Indigenous communities is required for a complete assessment encompassing all catch species that have cultural value.

vi. Other risk-based decision support tools such as Bayesian Belief Networks should be developed to incorporate the range of expert opinions normally associated with perceived risks, and to assess, evaluate and then communicate the effectiveness of proposed mitigation strategies.

14 General Discussion

The results of both “what if” assessments are similar in that the proposed FGARA development scenarios will on average, unlikely lead to the collapse of either fishery. However, there will be reduced fisheries catches, the scale of which will depend on the actual amount of EOS flow each river receives. Our risk models incorporate model uncertainties and, hence, provide a range of estimated reductions, with the mean impact all less than 10% of the historic mean catch in the three NPF fishing zones likely to be impacted. However, this may double to about a 20% reduction if maximum storage capacity is used. Whilst these reductions may seem moderate they are by no means trivial if sustained, particularly in the face of other cumulative risks to the fisheries not addressed here such as water quality issues associated with agriculture (Petheram 2013a,b) and loss of river floodplain habitat from sea level rise due to climate change (Bayliss et al. 2011). Additionally, the reduction figures in catch cannot be validated and are here meant to be used only as a guide to ‘what if’ development scenarios, not as prescriptions of exactly what will happen. They are only as robust as the underlying assumptions and uncertainties in the catch-effort-flow model. For example, small increases in fishing power and catchability of Banana Prawns over time can lead to significant changes in CPUE unrelated to abundance (Zhou et al. 2014). The correlation between Barramundi catch and total wet year flow volume would likely only hold for a run of favourable rainfall-flow conditions and may not be valid for predicting catch during years with successive low flow or prolonged droughts.

Balston (2007) suggests that changes in climate will have significant affects on the Barramundi and Banana Prawn fisheries on Queensland’s’ tropical east coast, and the same would apply to these fisheries in the GoC. Analysis of the Flinders and Gilbert EOS natural flow data since 1889, itself driven mostly by rainfall, suggests that decadal trends exist. This has implications for our understanding and management of freshwater and marine ecosystems because they are inextricably connected. Rodrigo et al. (2000) argued that decadal patterns in flow are likely because of decadal and multi-decadal climatic variability in the behaviour of global climatic systems that influence proximate drivers such as rainfall. For example, the Pacific Decadal Oscillation (PDO) is a pattern of Pacific climate variability with a similar regional climate signature as that for the ENSO (Latif 1998; Power et al. 2006; Nigam et al. 1999; Mantua et al. 1997; Zhang et al. 1997). PDO regimes may persist for two to three decades, whereas ENSO phenomena have an approximate period of three to seven years. In northern Australia we lack understanding of how climate forcing factors such as the ENSO and PDO may interact to influence tropical rainfall-flow events at decadal time scales. Early decadal rainfall studies (e.g. Power et al. 1999, Power 2006) suggest that the strength of the correlation between ENSO and precipitation may vary with the phase of the PDO. Bayliss et al. (2008) examined catchment rainfall and flow of several major streams in the NT and found approximate 20-25y periods for both, which was concordant with a similar period trend in the abundance of magpie geese on coastal floodplains monitored over 58y. These decadal trends were mostly explained (R^2=88%) by an ENSO-PDO interaction. Erskine et al. (2011) also found similar decadal trends in annual rainfall in the NT.

Decadal ‘regime shifts’ have been proposed for many fisheries (e.g. Beamish 1997, 1999). However, the effects of fishing and climate regime shifts over the same decadal time scales may be ‘confounded’ and so somehow need to be teased apart (Hilborn and Walters 1992). Walters and Martell (2004) state that one of the two main things we know so far about exploitation is the ‘general notion that fishing rates should be reduced during periods of low ecosystem productivity’.

In conclusion, rainfall induced river flow is a key driver in the Banana and Barramundi fisheries in the southern GoC, and reductions through water extraction for upstream development will impact negatively on the productivity of these fisheries. Mitigation strategies should aim to minimise downstream impacts so that the limited water resources are used for maximum social, economic and environmental outcomes. All mitigation strategies (see Part 4) should aim to maintain the long-term productivity of estuarine and marine ecosystems that are dependent on flood pulses for annual rejuvenation (see Junk et al. 1989 and Poff et al. 1997). This needs to be considered in light of decadal-scale changes in climate, which are currently poorly understood.

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Part 4 Mitigation

Rik Buckworth, Michele Burford, Rob Kenyon, Julie Robins and Peter Bayliss

Contents

Tables

Summary

In Part 4 we summarise the attributes of species and habitats evaluated at High and Moderate Risk levels, to identify the source of risk and to provide the basis of potential mitigation strategies. The principal sources of risk to species are: the decreased productivity, survival and growth that result from reduced flow; the consequent reductions in the extent, frequency and duration of brackish ecotones; the inundation of floodplains and saltflats; and reduced connectivity. Longitudinal connectivity could also be physically reduced by water storage infrastructure, such as dams and weirs, and ancillary infrastructure. We recognised that impacts of water development at the ecosystem scale might not be apparent from consideration of risks at the species, species groups or habitat scales. Mitigation strategies require that flows and connectivity are sufficient to support life histories and ensure the maintenance of catch (for fishery species). Availability of detailed information on the dynamics for some species, such as White Banana Prawns and Barramundi, means that mitigation strategies particular to them are more likely to be targeted. Mitigation measures for such species might also provide mitigation for other, lesser known species. We also suggest several mitigation actions that might be applied to dams.

Mitigation strategies should ideally be implemented within an adaptive framework (Walters 1997). Monitoring programs need to be deployed to ensure the effectiveness of mitigation strategies. A research strategy to close the most important knowledge gaps needs to be developed and implemented (see Part 5, Section 21). Lack of understanding of the effects of development at the scale of ecosystems should be addressed by developing ecosystem modelling to identify policy options, and to provide hypotheses about the potential impacts of water resource development in the Flinders and Gilbert catchments. These hypotheses would be used for testing and monitoring. Additionally, population-level understanding of key species in the Gulf of Carpentaria Inshore Finfish Fishery is required to better anticipate, mitigate and monitor potential impacts of future large-scale agricultural development.

16 Attributes of habitats and species at risk from development under the FGARA scenarios

The aim of this project is to provide information that will support decisions about future development of water resources in the Flinders and Gilbert catchments. These decisions should be informed with an understanding of the values of current ecosystem services supported by existing river flows. We have identified and described a set of species and habitats (assets) that are important to fisheries and conservation values, and/or are culturally significant to Indigenous people in the GoC. We have then evaluated these species and habitats in a qualitative risk assessment framework in order to identify the levels of relative risk posed to these assets by the FGARA development scenarios (Part 2). For two important fished species, white banana prawn and Barramundi, for which we have more detailed ecological and life history information, we did further quantitative analyses of the strong relationships between end of system flows (EOS) and catches (Part 3). In this section, we summarise the factors underlying the risks posed to the various species and examine potential strategies that might mitigate them.

Of 46 species or species groups evaluated, 23 were considered to be of Low or Negligible Risk, fifteen were evaluated as High Risk, and a further eight were at Moderate Risk, under the maximum FGARA water extraction scenario (Part 2). The attributes of the High and Moderate Risk species, in terms of flow dependence and habitat use, are summarised in Table 4.1. The species groups with the highest risk scores overall were all species that are highly reliant on estuaries and rivers to complete their life cycle (in full or in part). Of the four habitats evaluated in the qualitative risk assessment, two were considered to be at Low Risk, but we identified two as at Moderate Risk (Table 4.2; see Part 2, sections 8.36 to 8.39).

How would these risks arise? Water extraction will affect fisheries and species of conservation importance in a number of ways. In particular, seasonal or peak extraction during periods of reduced flow would impact on critical ontogenetic phases in a species' life history and have a disproportionate effect on survival or growth.

Floodplains

The species richness and the density of individual species on floodplains can be very high (Pusey et al. 2011, p. 78) and several of the High and Moderate Risk species are directly dependent on floodplains and saltflats, habitats found to be at Moderate Risk (Table 4.2). Excessive water extraction will reduce the scale and duration of flooding and inundation of floodplains and saltflats (Burford et al. 2010), and this in turn will affect food availability and habitats for fish and other critical species (Jardine et al. 2012a; Jardine et al. 2012b). Thus, water extraction impacts the interdependent sustainability of:

1. individual species, for example, through the loss of connectivity or environmental cycles that cue ontogenetic migration
2. the production and provision of habitat services, through the impairment of catchment processes.

The duration of flooding of habitats is important. Floodplains and saltflats both require inundation for periods of ten days to stimulate the algal growth that supports the food web for fish and other species that utilise these habitats (Burford et al. 2010). But the fish, crustaceans and other species that depend on these habitats typically derive benefits over some months of flooding and the duration of flooding needs to be sufficient to maintain species' life histories—a period of two to three months is likely to be needed.

Some species use floodplains for reproduction, a critical stage of their life history. In a major review of the freshwater biological assets of tropical Australia, Pusey et al. (2004) describe migrations from dry-season remnant habitat to ephemeral abundant wet-season habitats where they spawn and the juveniles exploit the flood-stimulated habitat productivity. The floodplain habitat sustains the populations of these species until after the wet season, when they return to the river channels and remnant habitats as adults.

Several additional species in Table 4.1—those that are dependent on estuaries—are indirectly dependent on floodplains and saltflats. Runoff from floodplains and saltpans is important for supplying nutrients to rivers, estuaries and coastal areas to promote productivity. Water extraction will thus affect the productivity of riverine and estuarine reaches of these systems by reducing longitudinal flow and runoff from floodplains and saltflats. Productivity studies in the Flinders River and adjacent Norman River have shown that these systems are limited by nutrient availability (Burford et al. 2012; Faggotter et al. 2013). Yet these rivers support high biodiversity (Pusey et al. 2011). Therefore catchment runoff is critical for providing the nutrients needed to promote productivity, and has flow-on effects to fish and species of conservation importance.

Salinity and nutrients

The effects of reduced flow on estuarine salinity and nutrient levels directly impact resident species. In addition to the loss of estuarine through-flow due to extraction, there can be loss of productivity downstream of dams due to trapping of nutrients and sediment. Most of the High and Moderate Risk species of Table 4.1 depend on the estuaries, at least in the juvenile phase. They rely on food sources within estuaries and these food sources are fuelled by nutrients from the catchment. In addition, resident juvenile banana prawn and mud crab in the estuary require low flow early in the wet season to reduce hypersaline conditions. Estuaries in the southern GOC become hypersaline in the lead-up to the wet season and in years of very low rainfall. Under such conditions, prawn growth may be inhibited until first rains and low flows reduce the salinity in the tropical estuaries to brackish levels. The growth and mortality of Banana Prawns is optimal in more brackish, wet season conditions, with temperatures around 28°C and a salinity of 25 (Staples & Heales 1991). In the case of juvenile Mud Crab, growth and mortality is optimal in warm brackish waters; about 30 to 32° C and salinity of 10 to 20 for growth, and 12 to 20 for survival (Ruscoe et al. 2004; Meynecke et al. 2010).

Connectivity

Longitudinal and lateral connectivity is a life-history requirement for many of the floodplain-dependent species in Table 4.1. Threats to connectivity, causing sections of the river to be physically disconnected when flow reduces or stops, are manifest through reduced flows via extractions and through the development of physical barriers such as dams, weirs, bund walls and vehicle access infrastructure. The Flinders and Gilbert rivers have a high percentage of migratory fish species compared with other rivers across northern Australia (Pusey et al. 2011). Additionally, iconic species are dependent on intact connectivity. For example, largetooth sawfish adults mostly inhabit coastal waters. Pupping, however, is in the estuaries and the juveniles move up into freshwater river reaches. They have been found 200 to 300 km upstream in both river systems.

Waterholes

Water extraction in the dry season is also an issue for waterhole-dependent aquatic species. During the low and cease-to-flow periods in the Flinders and Gilbert rivers (seasonally variable but may be much of the year—typically May to November especially in the Flinders River, and August to November in the Gilbert, based on median monthly flows; see Part 3, Section 13.3.2), aquatic species rely on both on- and off-channel waterholes as refuges. This includes adult Barramundi and the critically endangered Largetooth Sawfish. There are few perennial waterholes available to fish at these times and water extraction jeopardises the persistence of these systems. There is already a significant amount of water extraction in the Cloncurry River on the Flinders River system to supply the needs of grazing cattle, and further water extraction could result in a serious risk of substantial fish kills (Faggotter et al. 2010).

Cumulative Impacts

The most serious threat to fishery species and species of conservation importance is likely to be from cumulative impacts (USEPA 2003; Lindenmayer et al. 2012; MacDonnell et al. 2013). This includes the effects of changes to flow regimes from water extraction combined with those of dam construction, additionally causing alteration of sediment and nutrient loads. This will impact the interdependence of species' life cycles with the key facility provided by habitat. Additional to water resource development, there are many potential risks which could impose further environmental stress to fishery species and species of conservation concern. These include cycles in climate variability (e.g. El Nino/La Nina), long-term climate change, mining, grazing and invasive species. In a highly variable climate, potentially the greatest cumulative impacts could arise from periods of prolonged climatic-driven stress, such as series of dry years, as occurred in the 1960s, 1980s and 2000s. Seemingly small reductions in flow in average or large flow years, may have disproportionately large consequences if they occur in low-flow periods. In the case of Banana Prawn, a series of low-flow years typically produces poor catches and, consequently, poor economic performance (see Part 5, Section 22). Although the economic impacts of a series of poor catch years are probably similar for the Barramundi fishery, their reproductive biology means that prolonged reductions in flow will also lead to depressed recruitment. Although white banana prawn and Barramundi life-history requirements are relatively well understood, the same is not true for other aquatic species in the Flinders and Gilbert catchments. We can assume changes in flow regimes will have similar effects for many species, so that mitigation measures for Banana Prawn and Barramundi might provide mitigation for a range of other species. For this reason, we have examined Barramundi in detail but emphasise that the utility of mitigation measures as applied to other species will be species-dependent and more research is needed to examine mitigation measures for the lesser known species.

Consequences of flow alteration for Barramundi

The dynamics of Barramundi populations in response to flow are relatively well studied. As long-lived strategists (King & MacFarlane 2006), Barramundi are capable of persisting through poor conditions to immediately exploit good conditions. Strong recruitment pulses are a feature of several regional stocks of the species (Staunton-Smith et al. 2004; Halliday et al. 2012) and appear to support the productivity of associated fisheries for several years. Assessment of the water-use scenarios for the Flinders and Gilbert catchments indicated reduced recruitment (see Year Class Strength Analyses, Part 3, Section 13.3) and reduced fishery catches (i.e. CPUE with flow analyses). These reductions are likely to vary with climate variability, that is, where we are in the flood–drought cycle. In wet years, it is unlikely that water resource development in either the Gilbert or Flinders will impede very large floods (e.g. such as 2009) and the strong pulses in Barramundi recruitment associated with such events. Flow alteration is likely to have the greatest impacts where: (i) natural periods of low flow are extended both within a year (i.e. extending the months of no or low flow); and between years (i.e. extending the number of consecutive years with low flow e.g. droughts); and, (ii) natural periods of moderately low flow that, due to the water extraction, are turned into periods of low flow.

Although Barramundi is relatively well-known, there is little quantitative information on its minimum population sizes, or on recovery rates of populations from naturally occurring periods of low flow that are extended as a consequence of the proposed full development water-use scenarios. Thresholds are also poorly quantified. Sawynok and Platten (2011) nominated eight years as a critical threshold for the periodicity of strong recruitment events for Barramundi because of issues with the sex ratio of the spawning population. DSITIA (2013) nominated two critical thresholds for the periodicity of strong recruitment events of Barramundi: firstly, five years based on the average age of protandry (change from mature male to mature female); and secondly, 11 years, based on the longevity of Barramundi (Pusey et al. 2004). The extent to which these thresholds might be valuable in mitigation for other species is not known.

Table 4.1 Flow-dependent attributes of species at HIGH and MODERATE risk from water flow alterations under maximal FGARA scenarios, as indicated in Part 2. Abbreviations: MOD - Moderate; P - estuarine and marine productivity supported by flow is important to prey species; juv. - Juvenile; fw - freshwater

SPECIES	RISK	GENERAL FLOW DEPENDENCY	SEASONAL DEPENDENCY	ESTUARY USE	FLOODPLAIN USE	CONNECTIVITY
Largetooth Sawfish	HIGH	P; abundance related to flow		Important to juv.		Longstream required for ontogeny
Speartooth Shark	HIGH	P	Pupping Oct–Dec	Pupping site; important to juv.		Juv. use estuary and fw reaches
Northern river shark	HIGH	P		Important to juv.		Juv. use estuary and fw reaches
Freshwater Whipray	HIGH	P		Important to juv.		Juv. use estuary and fw reaches; ontogeny
Sea snakes	HIGH	P	Juv. present at the end of the wet	Important to juv.		
Barramundi	HIGH	P; catch, year class strength proportional to flow (mostly 1st year survival)	Jan–Apr is when most needed for recruitment and downstream movement in the Flinders		Main source of diet	Longstream and lateral to connect to floodplains and waterholes, catch depends on duration of connectivity
Pikey Bream	HIGH					Spawning migration, fw/estuary to river mouth
Sooty Grunter	HIGH	P; floods expand habitat for larvae and juveniles	Spawning requires T >25 °C, prior to/in wet season floods	May spawn on floodplain	Habitat for larvae, juv.	Movement from waterholes to spawning in ephemeral fw habitat
Blue Threadfin	HIGH	P; floods may cue reproduction		Preferred adult habitat; important to juv.	Juv. feed on floodplains	
King Threadfin	HIGH	P; enhances juv. survival	Juv. in estuaries in wet	Reproduction in lower estuary		
Mud Crab	HIGH	Recruitment strength related to strength of flow	High flows may temporarily reduce catch	Mating; juv. require brackish for growth, survival; adults use mangrove/mudflats		
Mullet	HIGH	Grow fastest in wet season	Seasonal rain and flow may influence downstream migration	Important to juv. and adults (prefer fw)	Production related to number of wetland patches	Spawning migration (fresh/estuary to marine); important to production
White Banana Prawns	HIGH	Low flow provides for optimum salinity	Jan-Mar flow stimulates emigration; minimal flow Jul–Dec enables immigration of juv.			

Bull Shark	HIGH			Important to juv.; adults may return to feed, pupping sites		Juv. use fw reaches
Migratory Shorebirds	HIGH	Flooding maintains productivity of mudflats etc.				
Green Sawfish	MOD			Important to juv.		Longstream required for ontogeny
Narrow Sawfish	MOD			Important to juv.		Longstream required for ontogeny
Mangrove Jack	MOD			Important to juv.		Larvae, juv. move into estuary, fw conditions. Need longstream and floodplain connectivity
Grunter	MOD		Aggregate near river mouth in winter	Juv. and adult feeding ground		
Black Jewfish	MOD		Form large aggregations Apr–Sep	Important to juv.		
Talang Queenfish	MOD			Important to juv., feeding ground for adults		
Blue Catfish	MOD				Seasonal feeding area	Required for access to seasonal feeding habitats (creeks, floodplains)
Winghead Shark	MOD			Feeding and nursery habitat		

Table 4.2 Flow-dependent attributes of habitats, as indicated in Part 2, at High and Moderate Risk from water flow alterations under maximal FGARA scenarios.

HABITAT	RISK	GENERAL FLOW DEPENDENCY	SEASONAL DEPENDENCY
Floodplains	MOD	Flooding needs to be long enough to promote productivity	Stimulation of benthic algae when flooded Release large amount of nutrients to the river and estuary when draining
Saltflats	MOD	Flooding needs to be long enough to promote productivity	Stimulation of benthic algae when flooded Release large amount of nutrients to the estuary and coastal zone when draining
Mangroves	LOW	Growth and survival enhanced by freshwater and nutrients	
Seagrasses	LOW	Flow provides nutrients in the long term but higher sediment loads have negative impact in the short term (increased turbidity)	

17 Potential mitigation strategies

A key consideration in mitigation strategies is ensuring that water management mimics natural variability (Poff et al. 1997; Junk 1999), that is, highly variable flow in the wet season, including first-season flows, and catchment-suitable periods of no flow in the dry season. Alterations to flow regimes must maintain variability and periodicity in keeping with the life histories of the species in the Gilbert and Flinders, and is a requirement of the WRP (Gulf).

Catches and abundance of several species listed in Table 4.1, and the productivities of most of the habitats in Table 4.2, are largely proportional to flow. Consequently, there will be trade-offs with flow extraction so that upstream extraction essentially will be traded off against downstream productivity. The overarching mitigation strategy for these assets would be to ensure sufficient flow for their life history to be supported, and catches maintained at levels that reflect current-use flow regimes. More complex mitigation strategies might be built by addressing the FGARA scenarios that provide more flow than the 'worst-case scenario' considered in Part 2.

Mitigation might also be addressed via other known attributes of the species. In the case of banana prawn and Barramundi, for example, the links between flow and species' dynamics are well understood, and thus banana prawn and Barramundi catches will be impacted by any extractions (see Part 3, Section 12.4), as they are largely dependent on flow (Table 4.1), as is the productivity of floodplains (Table 4.1; Table 4.2). Monitoring and mitigation measures for such species might provide mitigation for other, lesser known species.

Floodplain inundation

The duration of flooding on floodplains and saltflats needs to be sufficient to maintain species' life histories. Flooding of more than one week is needed for primary productivity to commence, and a period of two to three months is likely to be needed to allow sufficient time for fish feeding and reproduction; this will be species dependent. Mitigation strategies must accommodate not only the average recurrence interval (ARI) between flows of a particular height (e.g. to ensure floodplain inundation for a desired period), but also ensure that responses are contained in the strategy that will not further prolong already prolonged periods of poor rainfall, when the natural systems are already stressed. Although species may be adapted to the sequence of alternating dry and wet seasons, the variability of these seasons means that risks to species and habitats occur even within the extremes of natural flow regimes.

Wet season flows

Flows are most needed in the period January to March/April (Table 4.1). A mitigation strategy that ensures as close to natural flows as possible in this period, ensuring emigration of Banana Prawns to the offshore adult habitat and the survival and growth of juvenile Barramundi, is likely to protect this aspect of the production of these species, as well as the life histories of many others.

Longitudinal and latitudinal connectivity is vital for the successful completion of the life histories of several of the species considered in this study. Water-use scenarios need to ensure that flows are sufficient to maintain the catadromous life history of species, such as the Largetoothed Sawfish and Barramundi, and must not contribute to the decline of threatened and endangered species. Following the prolonged dry season, with its seasonal loss of connectivity and shrinking habitat and resources, the first flows of the early wet season ensure the survival and movements of many species. A potential mitigation strategy for catadromous (as well as diadromous/anadromous) species would include allowing early wet-season flows to pass downstream. The cumulative effects of a series of years of low rainfall (prolonged droughts) and thus low-flow years are also an issue.

Connectivity

Again, mitigation approaches must have mechanisms that support connectivity during summer months in prolonged periods of poor rainfall, but acknowledge that there are periods when there is naturally minimal flow. Additionally, barriers to the movement of fish and other species on floodplains should not be permitted (e.g. bund walls and vehicle-access infrastructure that blocks creek connectivity). One strategy for species such as Sawfish, where upstream access to freshwater habitats is critical, may be to limit human-induced connectivity constraints to as few reaches of the Flinders or Gilbert as possible.

The maintenance of natural flow regimes and engineered solutions to ensure longstream connectivity can mitigate the effect of dams on connectivity. These can be achieved by:

- off-channel storages in preference to on-channel dams as they allow fish passage longitudinally, maintain the natural metrics of low flows and first-season flows, and reduce the trapping of sediment and nutrients which lowers productivity downstream

- offstream water storage with benched offtake by its design does not impact the river channel continuity or connectivity. However, the water offtake is set into the river bank and the level at which the offtake becomes active is critical. The benched level of the water offtake allows environmental flows and low flood flows to pass the location of the offtake unimpeded. In years with low rainfall and runoff, 100% of flows may continue downriver. The offtake design has two features that are critical to the maintenance of a healthy estuary/lower river:
    - the long-term permanence of the benched offtake

    - the ability to block the offtake and retain all flood flows in the river channel when the offstream storage in full to capacity
- fish passages for on-channel dams to aid migration. However, researchers have limited understanding of passage design for tropical species.

Waterholes

Many aquatic species, such as Barramundi and Largetooth Sawfish, rely on perennial waterholes as refuges. One mitigation approach is to identify the key waterholes for these species and ensure sufficient flow to them.

Sediments and nutrients

Dams and landscape erosion also affect the delivery of sediments and nutrients downstream. Mitigating the problems created relative to these attributes should address the following:

- Offstream dam systems require protection from flood flows once they are full. Additionally, offstream systems can cause erosion, particularly during high flood periods if not well designed and managed. Sediments from erosion will impact on ecosystems downstream.

- Small dams are better than large dams as they trap fewer nutrients, needed for downstream productivity.

- Variable-level water offtakes from dams ensure that water is accessed from surface rather than bottom waters. This will reduce the negative effects on fish of releasing water with lower temperatures and dissolved oxygen, and larger amounts of dissolved nutrients. However, algal blooms in surface waters should be avoided as they can also cause low oxygen conditions downstream, causing fish kills.

Prioritisation of conservation areas

When preserving river, estuarine and floodplain habitat, there is a need to prioritise areas of greatest importance for fish and fisheries. New tools are available to optimise areas for conservation, for example, a study done in the adjacent Mitchell River identified optimal refugia for freshwater biodiversity (Hermoso et al. 2013). Additionally, techniques have been developed for inclusion of a temporally explicit component to conservation, which integrates water residence time into spatial allocation of priority areas (Hermoso et al. 2012). For Barramundi, several thresholds for periodicity of large recruitment events have been identified (Sawnynok & Platten 2011; DSITIA 2013). Mitigation strategies for the species could target a requirement for flows to be allowed to pass to the estuary if there were a specified number of consecutive years of flow. A tiered approach may be appropriate whereby a greater proportion of the natural flow regime should be required to pass to the estuary as the number of consecutive years of low flow accumulates to >5 years, >8 years, and >11 years. This may then provide various levels of mitigation for: (i) the commercial fishery and (ii) the Barramundi population.

18 Evaluation and recommendations for further work

Mitigation strategies should ideally be implemented in an adaptive framework. The strategies should include explicit means of gathering information on performance, and mechanisms by which the strategy and tactics applied can be updated to include new understanding. In the short term, a lack of adaptive capability needs to be managed by making the framework more conservative. The performance of mitigation strategies should be monitored, and the costs of monitoring and response included as part of the evaluation of potential new developments. Productivity of Barramundi and Banana Prawn, and other fishery species from the Flinders and Gilbert systems, that is lost as a consequence of upstream water use in the catchments will not be available to the fisheries—it cannot be caught elsewhere in the GoC. For impacted commercial fisheries, re-structuring (changes to licensing arrangements, perhaps even reducing the number of licences that can operate) might be appropriate rather than displacing fishing effort to other systems (Fisheries Queensland 2013). Costs arising from the restructure processes should be recognised in decisions about trading off upstream water use with downstream productivity.

In the case of the species that are most important to fisheries (commercial, recreational and Indigenous), the performance of mitigation strategies can be monitored. Catch and effort information is already collected for the commercial fisheries and recreational fisheries are regularly surveyed. There is an opportunity therefore to use and build upon this existing process, and adding further research and information collection explicitly for monitoring the effects of water management. For example, gathering information on year class strength in Gilbert River Barramundi before and after development would be informative. Studies in the downstream section of the Flinders and Gilbert rivers to elicit Indigenous cultural values of impacted assets are required both before and after the implementation of development to ensure that these values are sufficiently supported by mitigation strategies (see Part 5, Section 23). Similarly, the potential socio-economic impacts of losses to other sectors need to be quantified and evaluated (see Part 5, Section 22). More broadly, mapping of resource dependencies is required to provide likelihoods of different sorts of social and economic impacts, including valuations of dependencies and the ways that they might constrain mitigation options.

Crucially, ensuring that the impact of altered flow regimes can be evaluated would include information gathering before and after development (i.e. incorporating before-after-control-impact (BACI)designs).

For species that have conservation importance but about which fisheries data are likely to be uninformative, separate monitoring programs will need to be developed. Monitoring of abundance of important species might be appropriate for species that depend principally on the productivity driven by flow. For species that depend on longitudinal and lateral connectivity, such as Sawfish, mitigation strategies to ensure connectivity should be evaluated in terms of sustaining the viability of their movements.

In reviewing the life histories of the species in this study (Part 2), it was apparent that there were many knowledge gaps, particularly at fine spatial and temporal scales. More generally, there were many knowledge gaps on the distribution and abundance of several species in the Flinders and Gilbert catchments. Floodplain ecology and use by these species, spatially and temporally, is poorly understood. Despite the dynamics of Banana Prawn and Barramundi and links to flow regimes being sufficiently known to suggest more targeted mitigation strategies, the dynamics of other important species such as Mud Crab or Blue Threadfin are unknown. We are therefore unable to suggest mitigation strategies at finer time scales. We have poor knowledge of large ecosystem components, such as floodplains, saltflats and coastal areas (floodplumes) inundated by floodwaters.

We highlight the need to develop and implement a strategy to close the most important of these knowledge gaps.

Among these knowledge gaps is a lack of understanding of the effect of water development at the ecosystem level, which is not be apparent from our evaluations of individual species or species groups. For example, we do not know how system components (species, habitats) will interact under changed regimes. This knowledge gap should be addressed by developing ecological models, such as Ecopath with Ecosim and Ecospace (Walters et al. 1997; Walters et al. 1999), as a means of summarising available knowledge, and to allow investigation of policy alternatives. Outcomes will support the identification of alternative hypotheses that can be tested experimentally. While it was beyond the scope of the current project, an Ecopath ecosystem model of the entire GoC previously developed by Griffiths et al. (2010), and/or Bayesian belief network (BBN) probabilistic approaches developed in the adjacent Norman River system (e.g. Duggan et al. 2012) could be used for ecological modelling focused on the Gilbert and Flinders system.

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Part 5 Future research and monitoring needs

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20 Introduction

Part 4 of this report specifically recommends potential mitigation strategies to minimise risks to Gulf fisheries and ecological values that were identified in Parts 2 and 3, and during two stakeholder-experts workshops, and so will not be addressed here.

Part 5 specifically addresses Objective 7 of our project, and is a summary of the detailed recommendations made in Parts 2 (qualitative assessment) and 3 (quantitative assessment) of this report related to identified priority knowledge gaps and future research and monitoring needs required to fill those gaps (see Part 1 Section 3).

Objective 7. Identify priority knowledge gaps and monitoring and research needs to fill those gaps.

Whilst critical and substantial knowledge gaps in remote and isolated regions of Australia such as the Gulf of Carpentaria are inevitable, given the scope and nature of our assessment, there were also severe time constraints limiting what could be realistically achieved in 4-5 months. As a result our assessment does not address the full range of socio-economic and cultural values likely to be impacted by sustained reductions in flow for large-scale agricultural development, such as the scenarios presented in FGARA. Hence, we have included two additional sections in our Recommendations that outline in more detail future research needs that were flagged in Parts 1 and 2 of this report. The first of these additions is a preliminary quantitative assessment of the potential economic impacts on coastal communities and ecosystem services in the Gulf, indicating the need for more detailed analysis and incorporation of results into the risk models (Section 22). This would lead to a more robust assessment of the inevitable tradeoffs between agriculture and other market and non-market values (e.g. recreational fishing and conservation values). The second addition makes recommendations on the need for a comprehensive assessment of Indigenous marine and coastal values likely to be impacted by sustained water harvest in the region, and builds on consultations commenced in the catchments during the FGARA study (Barber 2013).

21 Knowledge gaps and priorities

21.1 Qualitative risk assessment for species and habitats

There is a general paucity of reliable biological and ecological information on marine and estuarine species in the Gulf that may be vulnerable to sustained reductions in freshwater flow. This necessitated the adoption of a qualitative risk assessment approach to underpin our overall assessment of potential risks to fisheries and ecological values. The qualitative approach is therefore a “blunt instrument” for water managers because it necessitates starting at the “worst case” scenario with respect to reductions in flow (i.e. not other potential cumulative impacts), and then working backwards to better define the likelihood of exposure. Hence, Part 2 of our assessment recommends strongly that detailed biological and trophic studies be implemented on a handful of carefully chosen key indicator species to support quantitative population and ecosystem level assessments (see Part 2, Section 9 for details). The two main recommendations to address priority knowledge gaps are:

i. Trophic studies to determine critical relationships between key indicator species and freshwater flow, particularly ecological thresholds in relation to the hydrograph if any; and

ii. Large and small-scale movement studies to better understand how changes in connectivity between marine, estuary, river and river floodplains due to reduced flows will impact on population level responses to key indicator species.

Closing these key knowledge gaps would allow more formal fisheries stock assessment methods to be used (see Part 3 for Banana prawns and Barramundi), which in turn would allow potential impacts on populations and associated social, economic and cultural values to be quantified and more accurately assessed. Such a targeted long-term research strategy could be implemented under the WRP for the Gulf, as part of any planned continuous monitoring and assessment program.

However, although stock assessment approaches may be useful for species of economic importance, they do not allow exploration of the whole-of-ecosystem impacts of reduced flows. Use of ecosystem models, such as the Gulf of Carpentaria ecosystem model used to identify keystone species in this project, is a powerful way to examine the broader ecological impacts of environmental perturbations such as a sustained reduced flow. These models are becoming increasingly flexible, with the capability of integrating hydrological models to drive ecosystem changes through specified flow regimes. They can also be spatially explicit, allowing different habitats and regions of the modelled region to be examined in isolation, which would be particularly useful in examining the potential ecological effects of introducing a range of flow regimes in only the Flinders and or Gilbert River, or in combination.

Part 2 also recommended improved species-specific reporting in commercial fishery logbooks if CPUE is to be used as an index of abundance for future population assessments, especially for potentially vulnerable shark species such as Hammerhead and Whaler sharks.

21.2 Quantitative risk assessment for Banana prawns and Barramundi

The main recommendation from the quantitative risk modelling of Banana prawn catch in the Gulf to sustained reductions in flow in the Flinders and Gilbert rivers is to initiate a finer spatial analysis of catch and effort data using the 6nm grids. This is necessary to determine the relative contributions of the EOS flow of all major rivers in the southern Gulf flowing into the adjoining NPF fishing Zones 7, 8 and 9, particularly Zone 8 with respect to the FGARA development scenarios for the Flinders and Gilbert rivers. Our models likely overestimate the catch attributed to the Flinders and Gilbert rivers to an unknown extent. This is likely to be proportional to the EOS flow of all other contributing rivers, particularly the Norman River as it is situated between the two. Other surrounding rivers that will contribute to total flow into Zone 8 are the Staaten River to the north and the Leichhardt and Nicholson-Gregory rivers to the west. Hence, whilst the combined Flinders and Gilbert EOS flow is likely a good relative index of the total EOS flow from all southern Gulf rivers, and hence the relative comparison of catch reduction between FGARA development scenarios is suitable, they remain positively biased to an unknown extent. Therefore, if possible, more detailed spatial modelling is required to apportion the true absolute reduction in catches after accounting for the flow contributions of the other rivers. Preliminary analysis of mean wet year EOS flow data (2030-2007) for these rivers using NASY rainfall-runoff models (Petheram et al. 2009a,b; pers. comm.) indicates that the Flinders and Gilbert rivers contribute 16% and 24% (40% combined) to total flow volume entering southern Gulf waters, respectively, whilst the Norman River contributes 22%, the Leichhardt 11% and the Staaten 26%.

There are two main recommendations from the quantitative risk modelling of Barramundi catch in the Flinders and Gilbert rivers to sustained reductions in flow, and that may also apply to other flow-dependent species in the Gulf Inshore Finfish Fishery such as King Threadfin and Mud Crabs. These are:

iii. Population level data is required to better assess potential impacts of flow extraction on Barramundi catch given the possible confounding of movements with in situ recruitment and survival in the catch and effort data. Year-Class-Strength (YCS) studies for Barramundi and King Threadfin Salmon should be continued in the Flinders and initiated in the Gilbert. Because they dominate the catch, they would make ideal indicator species for monitoring potential inshore fishery impacts from future agricultural developments such as those proposed by FGARA.

iv. The influence of the extent, frequency and duration of floodplain floods in relation to catch needs to be examined in greater detail for all flow-dependent species in the Gulf Inshore Fisheries. There may be possible low flow relationships that water managers can use as a Threshold of Concern (ToC, DSITIA 2013) and, total wet year flow volume may not explain all critical population processes of Barramundi, for example and, consequently, their abundance and catch.

In addition to the above technical recommendations with respect to key knowledge gaps in species population ecology and risk modelling, there were three other major recommendations made to strengthen future assessments and to support future monitoring.

v. The full economic and associated social impacts of sustained reductions in Banana prawn and Barramundi catch could not be fully addressed here in detail, but is a critical part of any assessment of water tradeoffs between agricultural and fisheries production (see next Section, 22). Additionally, the economic and non-market impacts of water extraction on Indigenous and recreational catch, and on conservation values, were also not part of this assessment (see Section 23). We therefore recommend strongly that studies be implemented to address these shortcomings to complement the overall assessment results reported here.

vi. Further consultation and engagement with key stakeholders (e.g. the NPF and Gulf Inshore Fisheries industry sectors; Indigenous, recreational and conservation interests, farmers and graziers etc.) is required to communicate these risks accurately with respect to the true uncertainties embedded in the assessments. For example, the qualitative risk assessments are benchmarked from “worst case” scenario and the quantitative risk models are simply “what if” scenarios underpinned by many untested assumptions in the predictive catch-effort-flow models used in Part 3.

vii. As flagged in (v) above, Indigenous cultural values of coastal and marine resources that may be impacted by sustained reductions in flow in the Gulf are largely unknown (Barber 2013; pers. comm.; see Section 23), although Jackson et al. (2011) has documented important Indigenous socio-economic values in relation to river flow. Comprehensive consultation and engagement of coastal Indigenous communities is therefore required for a complete assessment encompassing all culturally relevant flow sensitive species and habitats.

viii. Other risk-based decision support tools such as Bayesian Belief Networks should be developed to incorporate the range of expert opinions normally associated with perceived risks, and encountered in this assessment, in order to assess, evaluate and then communicate the effectiveness of proposed mitigation strategies.

22 Potential economic impacts of reduced flows on coastal communities and ecosystem services (Sean Pascoe)

The economic impacts associated with reduced water flows in the region are highly uncertain and are an area for further consideration. Water flows are likely to have two types of impacts –impacts on industries dependent on resources affected by the water flow (market impacts); and impacts in terms of loss of environmental services (non-market impacts).

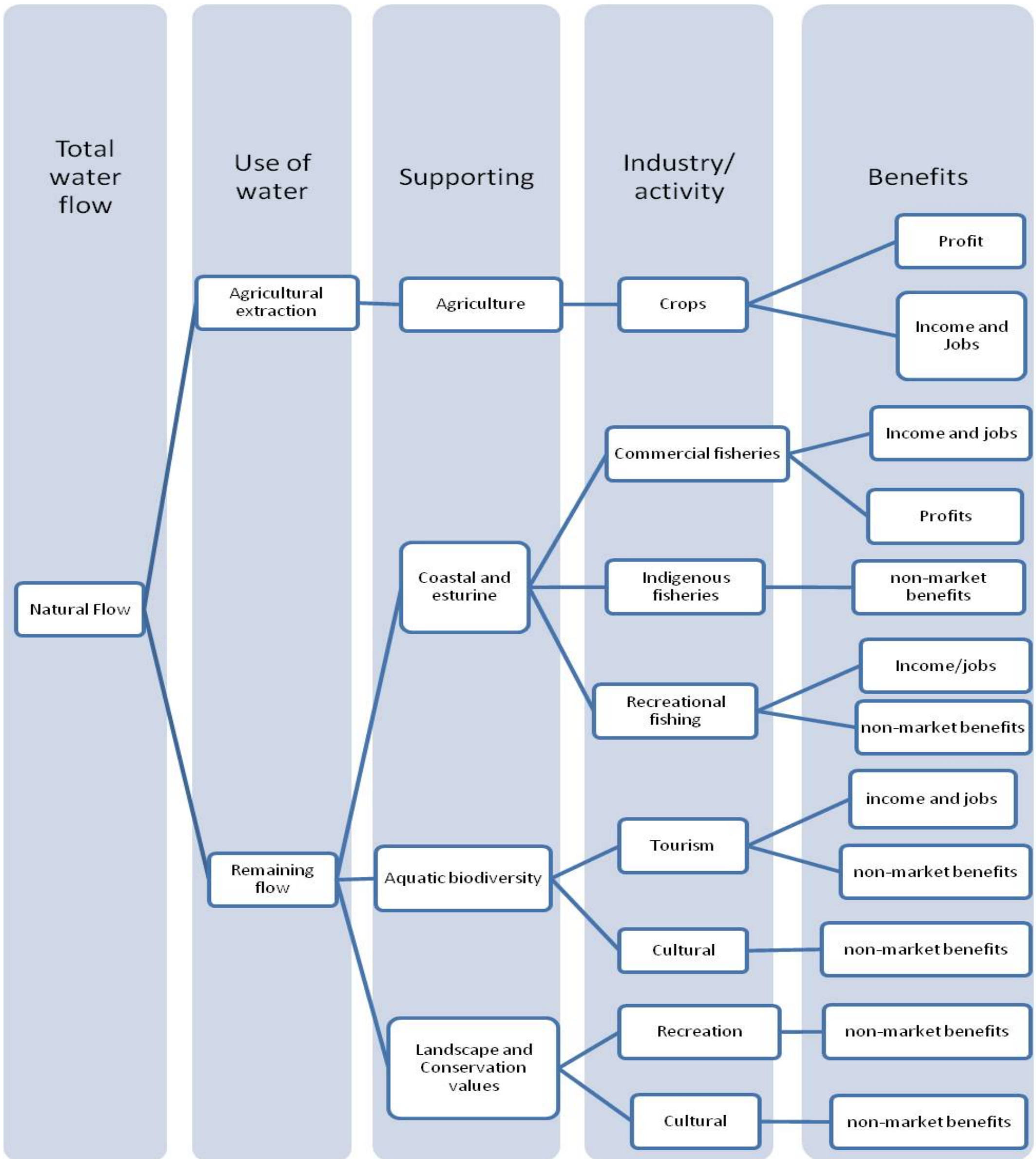

Figure 5.1 Benefits created by water flow

A summary of the key relationships between flow and the generation of market and non-market benefits is given in Figure 5.1.

Some preliminary assessments of the potential impacts of diverting water flow to agriculture are given below.

22.1 Market impacts

20.3.1 22.1.1 TOURISM

Recreational fishing is the primary reason for tourists visiting North-west Queensland (Stoeckl *et al.* 2006; Greiner and Patterson 2007). Nearly all tourists come from Australia (98.4%) (Greiner and Patterson 2007), The activity is characterised by a high proportion of retired couples (Greiner *et al.* 2004b; Stoeckl et al. 2006; Greiner and Patterson 2007). with fishing for food being a prime motivation (Greiner *et al.* 2013). This group accounts for as much as two thirds of the total catch by all tourists, with the key species being grunter, blue salmon and bream (Greiner and Patterson 2007).

The impact of reduced flow on the stocks for the key recreational species is uncertain. Similarly, the impact of changes in the stocks of these species on tourism in the region is also uncertain. Given that subsistence is a key motivating factor, and that elderly couples spend as much as 2-3 months on average in the area (Greiner and Patterson 2007), then a reduction in stocks is likely to result in a transfer of fishing effort by this group to other regions.

The impact of this on the local economy may be substantial. Tourism has substantial flow-on effects to the regional economy. Expenditure surveys of visitors to the region suggest that total tourism related expenditure in the region in 2004 was of the order of $11 million annually (in 2004 dollars), with an equivalent level of indirect benefits to the region (Greiner *et al.* 2004a; Greiner et al. 2004b). While the indirect economic impact (i.e. the derived economic activity generated in industries supplying the tourist industry) is roughly equivalent to the direct impact (the actual value of the tourism industry), indirect employment is thought to be as much as double that of direct employment (Stoeckl et al. 2006).

As noted above, the activity is characterised by a high proportion of retired couples (Greiner et al. 2004b; Stoeckl et al. 2006), with fishing for food being a prime motivation (Greiner et al. 2013). The retirees spend less per day, but stay substantially longer than the other groups that may individually spend more per day (Stoeckl et al. 2006). Loss of access to the fish resource is therefore likely to result in this group moving elsewhere.

20.3.2 22.1.2 FISHING IMPACTS

The impact of flow on the catch of banana prawns was identified as significant in the quantitative risk assessment. Although the degree of the impact is highly uncertain (as changes in flows are similarly highly uncertain), the risk assessment suggested that the impacts may be between 3 and 13% on average for the area affected.

The impact of this on the economics of the fishery is highly uncertain, as the contribution of the region to total banana prawn catches varies – significant in most years but less in others. A rough estimate of the impact can be made using economic information on the fishery published by ABARES (Skirtun *et al.* 2013; Skirtun *et al.* 2014) and using some simple scenarios. Using economic information on the fishery from 2007-08 (i.e. after the fleet restructuring had taken place), the impact of changes in banana prawn catches on revenues, costs and fishery profit can be estimated. The first scenario is that banana prawn catches are reduced by 8% on average. Rainfall and flows were above average during this period, while the preceding 6 year period was subject to relatively low flows (and lower banana prawn catches). The second scenario involved scaling catches down to their equivalent values based on the low flow catches in the preceding 6 year period.

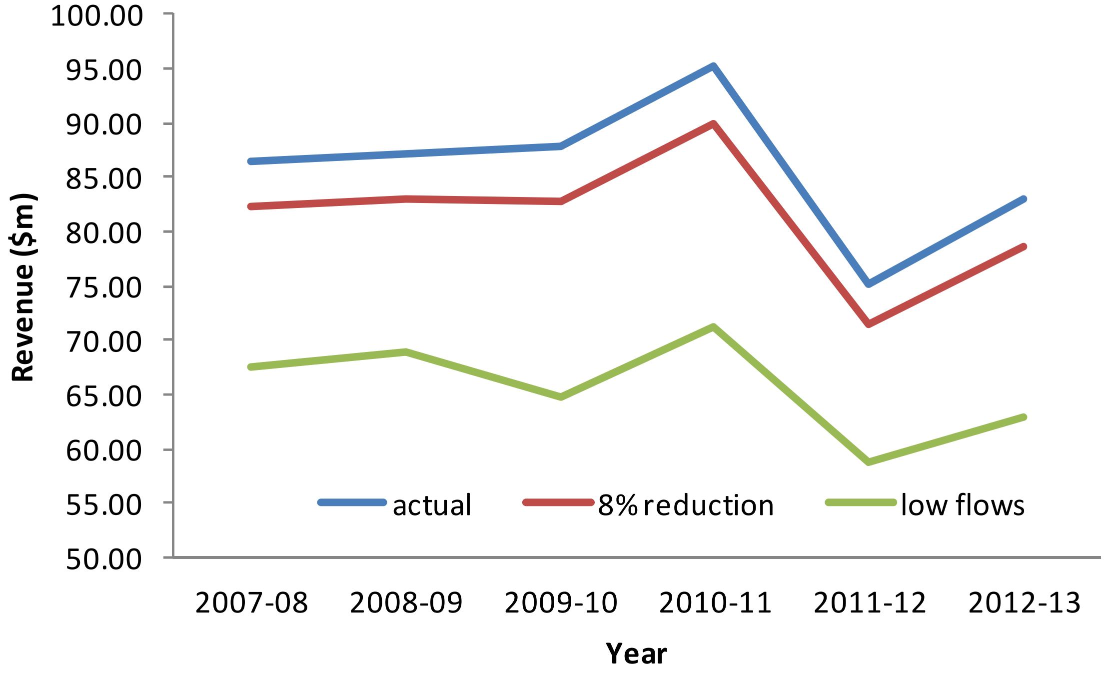

Figure 5.2 Estimated impact of reduced water flow on fishery revenues based on 2 scenarios

Changes in total revenues under these different scenarios can be seen in Figure 5.2. An 8% reduction in banana prawn catches over the last 6 years in the fishery as a whole would have reduced total fishery revenue by around $4 million a year on average. Sustained low flows (i.e. through extraction of greater quantities of water, particularly in good years) may result in reductions in revenue of around $20 million a year. The importance of the area affected to total catch varies from year to year, so these figures are likely to be overestimates of the total impact. However, they do suggest that the impacts may be substantial, particularly in years in which the area affected dominated the catch.

The impact of these changes in fishery profits can also be estimated (Figure 5.3), adjusting also crew costs (paid a proportion of revenue) and fuel costs (assuming fishing effort declined in proportion to catch). An average 8% reduction in banana prawn catch over the last 6 years would have reduced total fishery profits by around $2 million a year, whereas sustained reduced flows would have reduced profits by around $10 million a year on average, making the fishery uneconomic in some years and only marginally viable over the period as a whole.

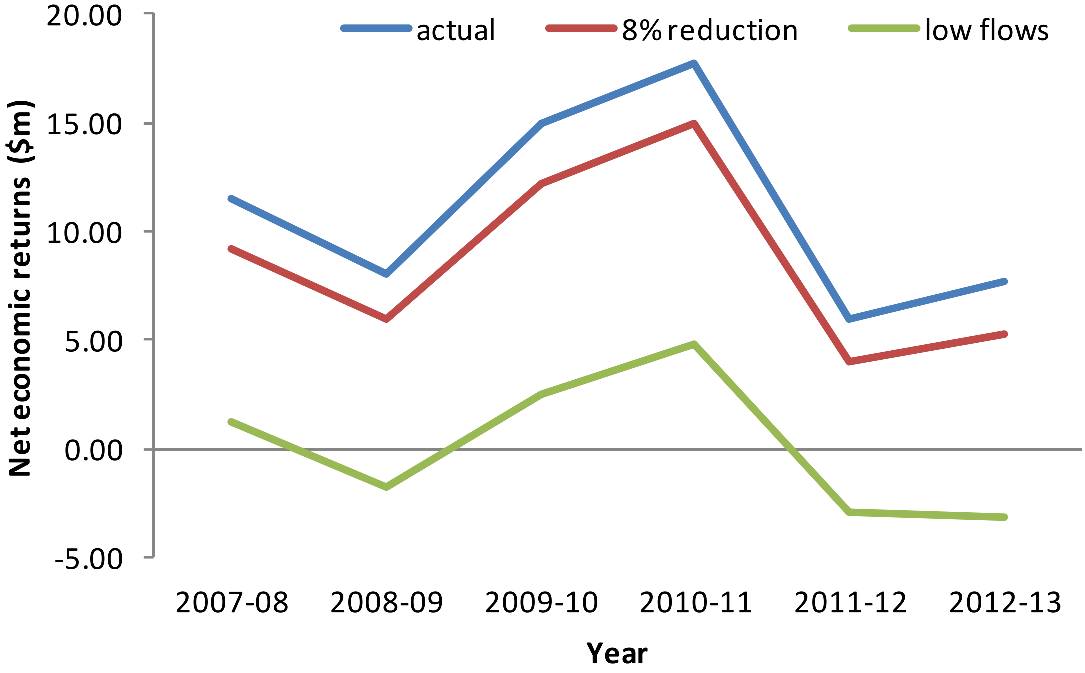

Figure 5.3 Estimated impact of reduced water flow on fishery profits based on 2 scenarios

These estimates are based on fairly simple assumptions so are indicative only. Factors that have not been considered is the impact of lower banana prawn catches on tiger prawn fishing effort, which may have reduced the short term costs but may also have had longer term sustainability issues. They illustrate that the impacts may be substantial, and require further detailed analysis.

The inshore fisheries such as the barramundi fishery are also likely to be affected. The risk analysis suggests that catches of barramundi may be reduced by around 7%. Economic data analogous to that used above for the NPF are not available for the barramundi fishery, so the potential impacts of reduced water flow on industry profitability and sustainability are uncertain, and require further investigation.

22.2 Non-market impacts

Non-market impacts are those that do not have an explicit price in the market place, and are hence difficult to value. While there is a wide range of non-market values, they may be generally considered in terms of use values and non-use values.

20.3.3 22.2.1 USE VALUES

Key non-market use values include recreational activities such as fishing and other use of the river system. While the market impacts of reduced recreational fishing in terms of reduced tourism to the region can be measures (as noted previously), there is loss in value to the tourists and other recreational users themselves from reduced recreational activities. Several studies have directly linked recreational values to river flow (e.g. Daubert and Young 1981; Duffield *et al.* 1992; Willis and Garrod 1999; Loomis *et al.* 2000; Ojeda *et al.* 2008). Several of these (Daubert and Young 1981; Duffield et al. 1992) suggest that recreational values of water flow may exceed the value of water to irrigation, especially in low flow periods.

The above studies have largely considered recreational values associated with direct use of the river system (e.g., for fishing, swimming or other recreational activities). Creel and Loomis (1992) examined the tourism benefits associated with maintaining wetlands, and found that the tourism value of water to maintain wetlands exceeded the value of this water for irrigated agriculture upstream.

Recreational fishing is likely to be the main recreational activity affected by reduced water flows in the Gilbert and Fitzroy river systems. How this will be affected by different water diversion strategies is unknown. Based on studies elsewhere, these impacts may be significant, particularly given the high costs involved in undertaking recreational activities in the region (indicating a high value of the activity).

20.3.4 22.2.1 NON-USE VALUES

The potential impact of reduced water flows in the system are uncertain, but are likely to include reduced habitats for birds and non-commercial fish species as well as other biodiversity impacts. These ecosystem services provide non-market values, the loss of which are often not considered when assessing financial viability of development proposals.

Studies elsewhere have found that the value of the impact on general ecosystem services due to reduced river system flows may be substantial. Loomis et al. (2000) found that agricultural development along a 45-mile section of the Platte River resulted in diminished ecosystem services such as natural purification of water, erosion control, habitats for fish and wildlife and recreational use. The loss of these services was valued at between $19 million to $70 million (depending on the assumptions of the values of non-respondents), far exceeding the cost of recovering these services from the agricultural sector (around $13 million). Around half of this value was attributed to use value (e.g. recreation), while the remainder was non-use value. Hanley *et al.* (2006) found that UK households are each prepared to pay £20 on average to improve river ecology from “fair” to “good”, as well as £21 to improve bank sides from “fair” to “good”. While this is not directly linked to river flow, it indicates that the non-market values of aquatic resources are potentially considerable. They also conclude that choice experiments are an appropriate technique to elicit such values (Hanley et al. 2006).

Within Australia, considerable attention has been paid to the Murray-Darling system, and the proposed management plan to divert water from agriculture to ensure minimum ecosystem services are maintained or restored. The development of agriculture along the river systems within the basin and the subsequent diversion of water from the system is believed to have resulted in degradation of 95 per cent of the environmental condition (Norris *et al.* 2001). Several non-market valuation studies have been undertaken to assess the potential benefits of restoring these ecosystem services. Using these studies, as well as additional choice experiments, Morrison and Hatton-MacDonald (2010) derived a total present value of these lost benefits to be in the order of $4 billion.

Most studies in this area have focused on the costs to agriculture of ensuring sufficient water flows to ensure minimum standards of ecosystems services are maintained. Qureshi *et al.* (2007) estimated that adequately maintaining environmental flows in the Murray-Darling system would result in a cost to agriculture of between $39 million to $75 million in terms of production, depending on assumptions about water trading. Dixon *et al.* (2011), however, suggests that the costs to agriculture in terms of overall profitability of ensuring environmental flows may not be substantial, as higher water costs are likely to be offset by lower land costs.

The ecosystem impacts of agricultural development in the Gilbert and Fitzroy river systems, particularly those downstream, have been identified previously as highly uncertain and areas for further research. Similarly, the value of these ecosystem services are also highly uncertain. Based on the values derived for the Murray-Darling system, these values may be substantial, but may therefore warrant further research.

22.3 Other impacts

The impacts identified above are only a subset of the potential economic impacts, and ones that can be readily identified (even if quantification requires further research). Other impacts that have not been captured, but will affect the region, include the loss of services related to these affected industries and the impact this can have on the local communities. Examples of these include potential loss of health services currently available due to the large number of tourists to the region (particularly elderly couples), potential loss of school places due to a declining population if reduced economic activity forced people out of the area, and potential loss of other services related to the fishing industry (e.g. fuel supplies, engineering etc). Accessing these services will be made more expensive to the residents if overall economic activity in the region decreases.

23 Indigenous coastal values (Marcus Barber)

As a distinct socio-cultural group within the regional population, Indigenous people have specific rights, values and interests that need to be accounted for in assessments of the potential benefits and/or impacts of water development. One component of the FGARA study was a catchment-scale assessment of Indigenous water values, rights, and interests (Barber 2013). This was based on a combination of consultations with key elders from Indigenous groups, information from the available literature, and additional analysis provided by water planning and heritage consultants (Jackson and Tan 2013; McIntyre-Tamwoy, Bird et al. 2013). The study emphasised the importance of Indigenous group-based planning processes, and of regional coordination, in providing foundations for effective Indigenous engagement with development proposals and for positioning Indigenous people to benefit from subsequent development.

Based on the research scope, the FGARA study prioritised consultations with individuals from Indigenous groups who were directly affected by water and agricultural development proposals on their traditional lands, and on the effects of such developments on the terrestrial, riparian, and aquatic environments of those groups living downstream. The impacts of upstream development on Indigenous coastal values were not extensively investigated, and therefore require further exploration. Also important to investigate in further evaluations of development options are the dependencies and interdependencies between Indigenous and non-Indigenous populations in the Gulf. As an important component of the regional Gulf population, Indigenous people may be directly and indirectly affected by the general market and non-market impacts of potential water development described above in section 22.

23.1 Indigenous hunting and fishing

Although the FGARA study was oriented to upstream and aquatic environments, there are clear indications in the existing data that fishing, hunting and gathering from coastal and estuarine areas was a crucial activity for local people. This activity had multiple bases, including the provision of food security and dietary diversity across seasons, as well as recreational, cultural and amenity value:

> *The two river systems join into one. That’s the main hunting area for us. The different seasons are there for different things. Just before the wet, it is the sharks, then the big white catfish and the cherabin come in the wet, and we collect heaps. There is plenty, there’s always a feed. In the wet season you can’t get wallabies, but the dry season that’s the time they fatten up. There are also the bush plants that come up around the river. Djungala, bush cucumber, comes up along the river. People love to go out there when the rivers are running.*
> *(Senior Gkuthaarn and Kukatj C in Barber 2013:41)*

Indigenous awareness of the significance of the coastal and estuarine environments extends to concerns about potential impacts on species that may not be targets for fishing and hunting effort, but are known to be vulnerable. This includes sawfish and bird colonies (Barber 2013:70). A range of flow-dependent species and/or natural processes may also be important to Indigenous people even if they are not specifically targeted for food purposes. For example, seasonal indicators (plants, weather patterns, etc.) are used extensively by Indigenous people to regulate the timing, duration, and extent of particular instances of hunting effort (Woodward, Jackson et al. 2012). These indicators are not fully documented from the lower Gulf, but may be sensitive to changes in flow regimes. Other flow-dependent species and processes may be particularly important for cultural purposes such as ceremonies.

With respect to water planning and water development, Indigenous consultation processes for the original Gulf water plan were considered to be insufficient by local Indigenous communities, but that process nevertheless highlighted the general social and economic importance of coastal and estuarine fishing:

The majority of traditional fisheries are concentrated on coastal and estuarine areas within two to four hours of travel by boat from communities. The fishing includes line fishing, crabbing, hunting dugong and collecting molluscs and crustaceans. The Gulf river deltas contain some of the Gulf of Carpentaria's most biologically productive marine areas. Reliance by Indigenous people on native food sources for subsistence continues today. The mangroves are a foraging ground for Indigenous groups in search of mud shells (large bivalves), crabs, carpet snakes and flying foxes. The fringes of the mangroves also provide large holes or hollows for native bees nests. It is estimated that subsistence production may account for up to 23 per cent of all foodstuffs consumed by Indigenous communities (Department of Natural Resources Mines and Water 2006).

While these statements support the significance of the activity, the figure provided is a general estimate rather than a specific figure related to the southern Gulf context. There is clear recognition amongst coastal Indigenous groups that currently there is insufficient data about the extent of Indigenous hunting and fishing effort. There is also insufficient data about importance of that effort, and of the returns it generates, to the status of local Indigenous social, economic, and health indicators.

Surveys of Aboriginal participation in fishing were meant to be undertaken as part of the Gulf of Carpentaria Fin Fish Management Plan but were never completed, and the plan is no longer in effect. The result of this situation is that, from a local perspective, the Indigenous fishery remains poorly understood and therefore potentially undervalued in current fisheries management arrangements. The lack of data also constrains the knowledge base and therefore the management capacity of local Indigenous managers, in particular the Normanton Rangers, whose area of responsibility encompasses the downstream sections of both the Flinders and Gilbert catchments. The Gkuthaarn and Kukatj peoples of the lower Flinders, and the Kurtijar people of the lower Gilbert, have both undertaken group-based planning processes in recent times and will soon finalise planning documents based on those processes. In both cases, additional research into the extent, cultural significance and economic importance of Indigenous fishing has been identified as a clear and immediate priority. It is envisaged that such research would be community-led and involve partnerships with and support from relevant agencies such as FRDC, Fisheries Queensland, CSIRO, and the Gulf of Carpentaria Commercial Fishermans' Association (D.Smyth, pers. comm.). Information derived from such a study would be very important in evaluating the impacts of changes in flow on Indigenous hunting and fishing effort.

23.2 Regional community dependencies and interdependencies

Indigenous effort is largely shore-based, and relies on simple fishing gear, meaning that the impacts on coastal species are of particular significance to that subcomponent of the overall Gulf population. However there is a strong awareness amongst Indigenous people of the important social and economic role of the wider regional Gulf community, and particularly of the commercial fishing industry. This is reflected in comments by individuals from upstream groups about the impacts of water development on estuarine, coastal, and marine fish stocks and the associated industry that relies on them:

The commercial fishermen may also have issues. Everyone is part of that country.
(Senior Ewamian B in Barber 2013:70)

We got fish, prawns,crabs in the Gulf. There is a big crab industry in the Gulf there. The water coming down affects them.
(Senior Ewamian E in Barber 2013:70)

Such impacts on economic activity in the wider Gulf may have a range of impacts on Indigenous people. They may be felt directly (e.g. through the loss or reduction of employment opportunities) and/or indirectly through the loss of key services associated with the impairment or reduction of commercial activity. The lower social and economic status of Indigenous communities also means that Indigenous people may be more vulnerable than non-Indigenous people to such declines in regional economic opportunities and/or service provision.

In understanding the potential for wider impacts of reduced flow, it is important to map and analyse the dependencies and interdependencies between different subcomponents of the Gulf human population, and how those dependencies relate to potential changes in the status of key species on which economic activity may rely. Impacts may vary geographically, demographically, according to cultural demarcations, or all three simultaneously. An impact that appears relatively small from a catchment or regional economic perspective may be critical at the local Indigenous community level. The cumulative social and economic impact on downstream Indigenous people of multiple upstream developments also needs further understanding. This includes multiple developments of the same type, and combinations of different development types, such as mining and agriculture. Different types of development may offer different risks and opportunities for local people, both Indigenous and non-Indigenous. Currently, the lack of knowledge about economic and social interdependencies within regional Gulf communities constrains the identification of development risks and opportunities, the evaluation of risk mitigation options, and the discussion about policies and processes required to make such mitigation measures effective. From the perspective of understanding Indigenous coastal values, rights and interests, research which addresses this knowledge gap is important to undertake alongside more focused studies of Indigenous hunting and fishing effort and of local Indigenous ecological dependencies. It provides important context for risk and benefit evaluations and decision making with respect to Indigenous people and water developments.

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Appendics

Contents

Appendix 1: Workshop 1 report

Assessing the water needs of fisheries and ecological values in the Gulf of Carpentaria

Identifying Important Species and Dependence on Flows for the Fishery and Ecological Values of the Gilbert and Flinders River Systems.

Project Workshop #2 15th may 2014

Rm 3.141 Queensland BioSciences Precinct, St Lucia

Workshop facilitator: Cathy Dichmont

The first purpose of the workshop was to capture the insights and values of stakeholders and experts in terms of the species and ecological values of the Flinders and Gilberts River systems, the second purpose was to provide Reference Group members as well invited experts and the project team members with an overview of the project's directions and methods.

Several presentations were given to provide both information and to stimulate discussion on the various topics; this report only briefly summarises those presentations but presents discussion in more detail, as recorded. A breakout session was held in the afternoon with the intention of eliciting input from the workshop members, information on the attributes of important species in the river systems and any other related information that they might be able to provide. Outputs from this session have also been recorded in this report.

Presentations were as in the attached agenda and are provided as part of the project materials.

The workshop was introduced by Cathy Dichmont, attendees introduced themselves.

Rik Buckworth provided an overview of the project's requirements, specifically that it was tasked to deliver:

"*A technical assessment of possible impacts to fisheries and ecological values in the Gulf of Carpentaria from key development scenarios specified in the Flinders and Gilbert Agricultural Resource Assessment (FGARA).*"

He emphasised that given the duration of the project, the workshop was designed to ensure rapid focus on those species and ecosystem elements regarded as important by the workshop participants. The short project duration also meant that the outputs of the project must essentially be qualitative, providing an indication of the sort of effects that extraction of water for irrigation might have, but would mostly be unable to indicate their magnitude. Additionally, an important aspect of the work would be identification of further work that would be required.

Cuan Petheram: The Flinders and Gilbert Rivers Agricultural Resource Assessment (FGARA).

Dr Petheram provided an overview of the Assessment – an evaluation of feasibility, economic viability and sustainability of irrigated agriculture (IA). The purpose of the two year Assessment, published early in 2014, was to evaluate to the scale of opportunity. It did not seek to provide advocacy, replace planning processes, nor assume a regulatory environment.

The project output comprised 16 technical reports providing underpinning for summaries of catchment reports, which are summarised further into summary reports (16p). The basis of the assessment was a suite of models, including biophysical, agriculture development, and socio-economic components. Case study scenarios were: pre-development historical climate, A- historical climate and current development, B- historical climate and future irrigation development. Case studies were all based on scenario B to evaluate scale of opportunity for IA.

A river system model (RSM) was developed for each catchment, making it possible to evaluate water ultimately reaching the Gulf of Carpentaria (GoC). RSMs are suitable for simulating regulated rivers as they have the capability to incorporate dams, irrigators, extractions, environmental flow provisions and a variety of complex management and operational rules.

Case studies

Case studies were developed to illustrate how the different components of the Assessment can be put together. Case studies were geologically situated based on three of the more suitable water storage options in each catchment.

The Flinders catchment is largely flat and areas of higher relief are often geologically unsuitable for large dams. While in-stream dams could be constructed in a number of locations, but they would be expensive and have low-yields as a result of relatively small inflows. Water harvesting and storing water in farm scale off-stream storage appears to be the most logical approach.

Cave Hill: potential to irrigate 12k ha sorghum. There is a long history of interest in developing Cave Hill dam, and it appears to be one of the better options in catchment but requires further geotechnical investigations.

O'Connel Creek: long history of interest, water diverted to storage from Flinders River. In this case study water is used to grow rice planted in January and February

Water harvesting and irrigation mosaics are thought to be the most likely form of irrigation development in the Flinders catchment, with farmers diverting water to store for irrigation of a variety of short season crops.

The Gilbert River has fewer existing dam studies as a result of less interest in irrigation and without the long history of people championing irrigation development as is the case in the Flinders catchment. The Assessment reviewed all existing dam studies and identified new options

The existing Kidston dam is disused. It was developed by a gold mining company. It is existing infrastructure but would require modifications to allow water to be released for irrigation. Additionally, the capacity of the dam is small, so it can only irrigate small area. Increasing the dam height would increase the capacity of the dam by 5 GL. The nearest contiguous area of soil is the Einasleigh common, about 40 km downstream of the dam.

Green hills Dam case study assessed a 12k ha cotton/peanuts/sorghum in rotation

Dagworth and Green hills Dams irrigated sugar cane, 30k ha.

The Assessment sought to provide information in a form that enables the reader to answer their questions, and not be limited to the case studies. The case studies are only meant to be illustrative and not necessarily the most likely form of irrigation development in these catchments.

Discussion (questions and answers and comments about the systems)

Q. what is the % extraction in the FGARA scenarios? A. This is difficult to discuss in simple terms as flow is so variable. e.g. Flinders: Mean flow 2500 Gl, median 1250 Gl. These figures don't capture lower catchment information, e.g. coastal floodplain inundation.

The case studies were all done individually, only one case study has two dams but there is potential to consider multiple scenarios. Land suitability assessment, too, was an important consideration– locating the best soils e.g. alluvial vs deeply weathered tertiary soils with high density of nodules -. Land suitability modelling was included in the FGARA reports.

For coastal floodplains, extraction in dry years will amplify dryness, with less impact in big wet years, as extraction would be small compared to flows.

The two rivers have very different catchment characteristics:

Flinders catchment – largely flat, underlain by rolling downs, small areas with elevated topography could site water storages but they are small and shallow. Also large variability in flows will influence irrigation. 70% of the catchment has class 3 soils – (i.e. moderately suitable with considerable limitations) - so large areas are suitable for growing, but there is also a salt risk with the rolling downs group. Soil is not a limiting factor. The median discharge is 1250 GL. There is not much flood area in the upper catchment. Waterholes in this system are very turbid.

Gilbert Catchment – undulating in upper reaches, igneous rocks, lots of potential storage sites, not much in way of alluvial soils. Considerably less soil, 2 million ha of class 3, but a large portion are deeply weathered tertiary sediments; there are other areas of reasonable soil - lots of small pockets - but not much contiguous area. Development is likely to be constrained above the 'neck' of the catchment. This river has more water resources than the Flinders River, with extensive floodplains in the lower reaches. There are waterholes that are persistent with clear water, due to the rocky country. .

Blaber: What is the potential for connections between river catchments during high flow periods e.g. between the Flinders and Norman catchments?

Petheram: RSM are simple models, and don't model movement of water across the landscape. However, the team has also developed hydrodynamic models which show this landscape movement. Complex models, they take a long time to run, so they were only used for specific events e.g. 2 week flood events. They were used to develop relationships between discharge and inundated area. They can be added to river system models. Tides are included as a boundary condition.

Q.: Did models consider years of high and low flows? A. Hydrodynamic models only covered specific flood events, but you can apply output to the RSM. The Qld government is using these relationships for water resource plans. Length of drought not longer than in south-eastern Australia but the magnitude of droughts is a lot greater. There are decadal pattern but based on 121 years of rainfall data, other than the annual cycle, the patterns of rainfall do not occur with a regular periodicity that is more statistically significant than white noise (i.e. that we can stochastically generate using an ensemble of models).. Future climate scenarios were not a huge driver for project.

Blaber: What about flow effects on vegetation? This might have indirect effect on fisheries. Petheram: RSM considered this in the context of waterholes.

Gary and Claudine Ward: In June you can walk across mouths of these rivers, they have very low flow. The Gilbert River tends to be a lot clearer, but gets more rain on the coastal section; low pressure systems influence this. Extraction timing would be important, e.g. presently heading to a low flow period. How will series of drought years influence flow and extraction impacts?

Robins: There are more fisheries in the Flinders River, but fishermen to tend migrate between catchments, using the Flinders as a transit river. The Gilbert is very shallow. Both are heavily fished for crab.

Ward: Have had three flood years then two poor / dry years. Barramundi abundance is very low at present. There has been very little prawn in the Flinders R this year.

Pillans: Gilbert R – There are historical records of sawfish in upper reaches; little known of this but work in WA shows relationship between flow and abundance.

Sawynock: There are 8-9 thousand tag records from these catchments; the data is available. Also can model fish movement information from the tag recapture data. The majority of barra don't move far in these systems - mostly <100km, but quite a few fish show large movements between rivers. Large movements are probably influenced by high flow events. Threadfins and barred javelinfish are also of interest to recreational fishers. These don't appear to move far but there are few long term data. King threadfin is generally system-specific but show rare long distance movements. Pikey bream don't move, they are restricted to small areas of habitat, and are short lived. The data can be used to produce GoogleEarth images showing tagging over time.

Dave Milton: There are large populations of migratory shorebirds in the region, 30 species of migratory and 16 species that are internationally important.

Note that connectivity among catchments is important for many species.

Shane Griffiths: Identifying important species

Shane described the importance of the area to fisheries and also briefly described the data sets that are available to provide insight into important species for the Gilbert and Flinders catchments.

In the National Recreational Fishing Survey of 2001, there were an estimated 77 000 fisher days spent in the area, landing around 400,000 fish and crustaceans.

The area is important for commercial fisheries as well. Consequently there is an extensive NPF and QDAFF logbook data set, CSIRO research information from regular surveys of the NPF and data from research studies in the Norman River, QDAFF scientific observer data, summary data ERA.

Shane noted that indigenous people were poorly represented in the recreational survey and the data were correspondingly poor. Worthy of note was that 70% of indigenous fishing is from the shore while there is a high rate of boat use by non-indigenous people.

Rob Kenyon: Conceptual models - life history of key species

Rob described the methodology of capturing important attributes of life history for important species by using conceptual models. The life history events and their timing can then be compared to the attributes of various flow scenarios.

He illustrated this with a conceptual model of banana prawn life history, in which adults (spawners) eggs and larvae are marine but post-larvae- juveniles enter the estuaries and grow optimally in water of reduced salinity (~25 ppt). Flooding is a cue for emigration of pre-adults to offshore. Nutrient plumes may also be important for productivity off shore.

This life history raises several questions about the effect of water extraction and its mechanisms: would impoundments negate low flow (creating connectivity issues), lead to stratification and the deposition of sediment within the estuaries?

Discussion point: Kennard: There are issues of changes to low flows and also barrier influences of dams for species migrations about half of freshwater fish species in the region need connectivity.

Shane Griffiths: Building risk assessments

There are several accepted risk assessment approaches. We are using a qualitative risk assessment procedure, basing our analyses on the Fletcher (et al. 2005) approach. This is a consequence-likelihood analysis –which utilises ranked input of the likelihood of an impact occurring and the consequences of that impact (also ranked). This Australian–standard compliant approach is based on expert opinion and scientific information when available. Examples were provided.

Discussion points: there are likely to be many species that are considered important to various degrees and, given a variety of scenarios, it is probably important to address the most extreme scenarios first, to establish whether there is a risk or not.

Kennard: Justification of risk, i.e. recording the mechanism by which ‘risk’ was captured, will provide the evidence for this and links to threats e.g. changes in late season flow.

It is important to be clear and explicit in assessments of risks.

Buckworth: Would impoundments completely negate low flows in some years? Cuan: yes, particularly if there is a series of low flow years, but it will depend on operation of the reservoir – this means that possible mitigation approaches could be explored that include variation in operation of the reservoir.

Suggestions: Include weighting of evidence in risk assessments e.g. the amount of information available, expert knowledge.

There were strong comments that we should include economic assessments, as it would help to inform policy decisions and that politically the wider irrigation development discussions will be about economic tradeoffs. Darryl McPhee has undertaken work across Qld which might inform the project as well as work by Jackson etc. Even “ball park figures” for economics would be better than no numbers at all.

Team members responded that economic analysis will be flagged as a recommendation, it is unlikely to be covered thoroughly in this project, due to time and resourcing issues. Assessment of recreational fisheries is likely to be especially challenging. We will attempt to engage Sean Pascoe in some preliminary analysis.

Claudine Ward asked whether the work by Lou Williams, relating barra abundance to rainfall, was available. Julie Robins indicated that she will try to track this down, but it was not really written up. The work found that catch and flow are related but varies among catchments and fishing effort was a major influence. Questions from this included whether flow drives catchability (lags) or recruitment (barramundi recruitment would have a 2-3 year lag). Barramundi are more challenging due these lags, compared with prawns with their one year life cycle.

Peter Bayliss, Melissa Duggan & Mark Kennard: Building risk assessments – conceptual models to desk top assessment and Workshop 2

Peter began with the point that “Good risk assessments are guided at the outset by good conceptual models” (Mark Burgman 2005).

He illustrated the construction of a quantitative risk analysis with a study of the effect of wet season water extractions on for the Daly River (NT) for 1983-2013. Barramundi catch was the sociological and economic endpoint and barramundi abundance was the ecological endpoint. Peter provided a conceptual model, and fitted the relationship of catch and catch rates (abundance index) with flow, and so developed a predictive model of the effect of reduction in flow (via water extraction). This was then used to build a Bayesian Belief Network (BBN). A BBN is a diagram for conceptualisation of risk, but is also predictive. It is capable of including both objective data and subjective expert opinion.

Bayliss concluded that the barramundi catch rates and abundance were both very sensitive to water extract, and with both recreational fishing and agricultural demands increasing, tradeoffs between recreational fishing and agricultural water use will need to be made. Bayesian Networks may have a role in negotiating an acceptable balance between competing water uses.

Peter displayed a regression for banana prawn catches from the Gulf of Carpentaria (minus the Weipa area), indicating that 73% of the variation was explained by modelled wet season flow volumes. He indicated that this could be the basis of a quantitative risk assessment for the project.

Mark Kennard: Environmental water requirements and ecological risk assessment of fish in the Daly River

Mark illustrated a semi-quantitative ecological risk assessment framework for the Daly River, NT.

Different extraction scenarios were described showing the impact of water extractions on dry season flow. The study combined indigenous and scientific knowledge and hydrologic habitat analyses but not risk assessment per se, were used to build a ranking of relative risk by (40) species. Conceptual models of impacts of dry season extraction e.g. impacts on depth and velocity of riffles were a key component.

This allowed the identification of focal species and quantitative risk assessment for those species, e.g. black bream and barramundi, and incorporation into BBNs. They also generated time series of habitat availability under flows.

This TRACK research is described in the book by Pusey et al. " Aquatic Biodiversity in Northern Australia".

Breakout Groups Session

Each breakout group was charged with the same set of questions

Q1 what is important to you: species, values, habitats, ecosystems?

Q2 what mitigation ideas do you have?

Q3 what info do you know of or could offer that could help?

Q4 what do you want to know from this project?

Fisheries Group

Q. 1

- Species in the commercial and recreational fishing in Gulf, incl. indigenous

Also catch:

- Banana prawns
- Barra
- Threadfin
- Mud crabs (big wet = small catch) – boost in seen 2 years after
- Salmon
- Bream
- Sea Perch
- Grunter
- Golden Snapper
- Black Jew
- Mangrove Jack
- Golden catfish (good product to sell + always in demand)

  Also consider
- Health of fishery (etc. habitat)
- Entire marine systems + linkages

NB: The Group liked Shane´s slides

- Social, economic values
- Feel catch days for recreational fisheries are underestimated
  - Grunter take is particularly understated
- Greasyback prawns: important because barra feeds on them
- From experience: Fresh flow keeps jellyfish numbers down.
- Fisheries (including recreational fishing v. important to towns like Karumba – e.g. Clinic opened due to increase in recreational fishing visits and other services as well)

NB: Barra a valuable fish as can sell flesh + other parts of animal

- Expansion of community very important; concern services facilities + community will suffer if fishing declines.
- Big wet year catches sustain fishers through bad years. Need the big years.
- Timing of water extraction important – different impacts from different timing.
- Fisheries adjust to changes in flow (low years + high years). However agriculture have little capacity to adjust to low years. Agriculture will require water regardless
- Little trust in authorities that they will only extract to a certain prescribed level.
- Political pressure will win in years of low flow.

Q2. Risk minimization

- Offstream capture/storage preferable to instream barriers
- Timing of extraction to be managed, e.g. a mitigation strategy could be to only extract during big wet years
- If demonstrated that fisheries are badly affected by extractions, agriculture should pay for re-stocking
- Science is available that reduces risk
- Monitoring systems need to be put in place
- Fisheries should be asking for offsets
- emphasised <u>MORE RESEARCH</u>
- MacArthur river mine. : community trust put in place by mine for "good works". Should be something like this in this scenario.

Q 3. Indigenous Rangers could help monitoring

- In low flow years:
    - Sick fish – impacts sale price + difference of catch that are able to be sold
    - Low prawn catches

July – Sep: Barra numbers drop so boats move to Weipa or Switch target sp.

- Definitive quantitative information on impact – reducing uncertainty + risk
- Needs of all stakeholders considered. Feeling at the moment is agriculture is rolling over the top of the other stakeholders, and once the agricultural systems are in place there will be no going back.
- Tagging program being put together (is low cost – Sawynock) – to collect baseline data for <u>before + after comparisons</u>
- Log book program – Gary
- Recruitment becoming a focus for data collection, + not just for target sp. – Bill
- Before /after comparisons will provide valuable info to make an argument
- Ord river issues should be studied as a comparison.
- Fisheries monitoring info
- Need info on flow in mouth (Hydro modeling only goes to last gauging station which could be well upstream)
    - TRACK study GIS data - perhaps Michele Burford might be able to provide?
    -

Q 4.

- What will be the major impacts?

- What will be the worst case scenario?
    - What are the fisheries going to lose (social + economic)
- Assessment of different impacts of taking water at different times
- Assessment of different impacts of cumulative impacts of taking from low flow years
- Assessment of the impacts of extraction on floodplains at river mouth.

Biodiversity + Habitat Group (Steve, Elvira, Richard, David, Shane and Karen)

<u>What is important?</u>

Maintenance of:

- Biodiversity, Ecosystem integrity/function
- Risk assessment species lists e.g. Iconic species – barra, sooty grunter, threadfin salmon, TEP – toothed sawfish, waders, sea snakes, est/fw crocs, Keystone – sergestids, *Macrobrachium*, mullets, bony bream, *Thryssa* sp, Endemics – gulf snapping turtle, Habitats: seagrass, mangroves, floodplain waterholes, riparian veg, intertidal mudflats (for all data and models on effects of turbidity and flow)

Q2: i. Minimum wet season flow to maintain connectivity, ii. First flush volume and velocity (FGARA identified as ecologically important), iii. Establish flow rules to maintain ecosystems

Q4: i. How can results be applied to extraction plans? ii what are wider Gulf impacts, as well as local extinction threats?

Connectivity + other system effects (Rob, Mark, Cuan, Julie)

<u>What´s important</u>

Workshop – Connectivity and other system

(Goc Fish.)

I process   " Production"

- Processes not just species

- Shape of hydrograph:
  - Wet, dry and steep change periods
  - different types of alteration – benched offtakes
  - Generic alteration – suggest 6 scenario changes to hydrograph to be assessed?
- Upstream – within a year

Flinders – water harvesting – broad recommendations. ie. Let first flow past after the Dry(eg in Fitzroy the first flow goes untapped as a as trigger for ecology),

To help assist risk from Δ`d hydrograph

- What h[illegible]y Darling, Fitzroy and other systems?
- Connectivity over > 1 year antecedent conditions connectivity frequency e.g. how important is a big wet after a series of dry years?
- Downstream to upstream connectivity and also lateral connectivity

<u>Indirect Effects</u>

- Barriers – roads, weirs
- Transformation of floodplain habitats
- Weeds, feral animals
- Nutrients, toxicants – other stressors – John Brodie

<u>Which bits of the floodplain are important?</u>

- Make sure ecological processes are captured

<u>Mitigation Ideas</u>

- Connectivity – letting flow past after a long dry (e.g. in the Fitzroy the first flow is untapped as a trigger mechanism for ecological events)
- Benched offtakes maintain low at base flows
- Maintain salinity gradient in lower reaches
- Risk of decoupling flow and other cues eg temperature, lunar phase
- Environmental flow releases if build dams.
- Strategic Assessment benefit trade-off – ecological values * costs – pre-emptive strategy
- Fish passage devices
- Identify Δ`s to distribution of species

- Offsets
- Minimise risk by concentrating effects on a restricted number of channels i.e. leave some 'pristine' channels in system
- Ideas from Murray Darling

Information to help

See

- Track website
- NAH NERP
- Indigenous values – Marcus Barber

What do you want to know?

- How will information be used for management?
- Will outcomes be publicly available?
- Will underlying data be available?
- Knowledge Gaps
- Quantitative Outcomes of Economic Tradeoff (impacts)

Social, economic and cultural values group

1. What is important to you?

- Reduced flow rates in rivers (barra, prawn)
    - More fishers out of work

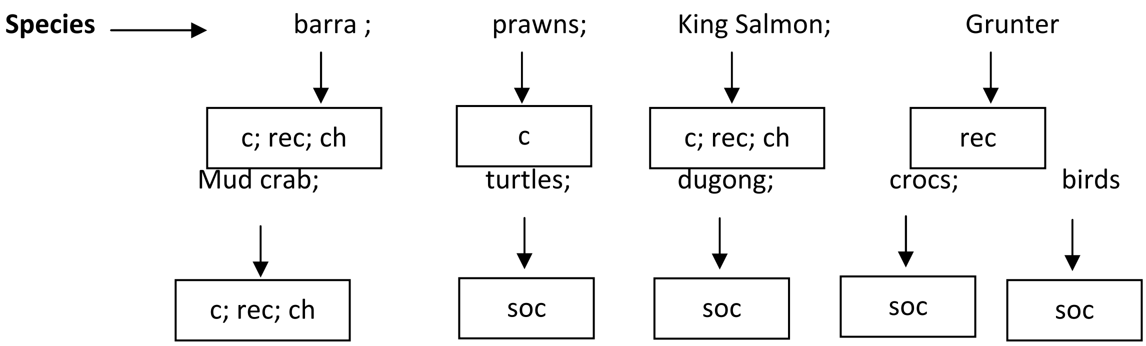
      ↓

      Less families

      ↓

      Less schools; clinics

- Economic + social impact on the fishery
- Loss or changes to ecosystem services (eg. Seagrasses ; mangroves)
- Tourist industry Karumba – Gilbert , Flinder, van Diaumans
    - how will timing change;
    - Access to Flinders at large high tide
- Bush Tucker
- No indigenous base at lower end of Flinder (cattle stations)
    - Native claim – Norman to Gore Point
    - Potential IPA for whole Gulf
    - Declared IPA west of Gore Point to NT borders
- Seagrass area in front of Karumba, maybe be indigenous hunting of turtle and dugong here



2. What mitigation ideas do you have?

- Control the types agriculture → affects water needs + water quality; consider minimizing evaporation loss
- Control timing of extraction
  - Not at low flow (ensure minimum - flow)
  - Take some in floods years
  - Big early wet? * might not be time for all spp.
  - Early part of Wet is not important for the prawn fishery
- Storage
  - Not ground water? (salt) ?
  - Evaporation?

    ↓

    a system that considers minimising evaporation
- Equity with existing
  - farmers, fishers
  - rights

3. What information?

- Link between flow + habitat/species abundance + then * social; cultural + economic values
- Economic value of :
  - a) commercial fishery (ABARES)
    - GVP (Lew Williams)↑ marine parts
  - b) Tourism – most rec fishing through tourism – (Carpentaria Council) – ABS
  - c) Agriculture
    - Cattle
- Variability of economic value – good catch / flow

  Good years in the fisheries allow for major upgrades etc. providing interannual links
- Population + growth
  - + seasonality (boat crew not local + depends on wet season)
- Need to talk to indigenous people for indigenous cultural values.
- Local fishing contests; seafood festival to be maintained as locally important
- Indigenous values

4. What do you want to know from this project?

- Is there any economic benefit from FGARA + others; flowing to the community?
- Flow → spp/habitat → social; cultural + economic

- Impact of building dams/infrastructure + then it doesn´t work
    - Legacy effects

Summary

A point of emphasis from the workshop was :
"If economics information is coming through as needed, can we include it?"

Terrestrial/marine knowledge integration is important

There was consistency in the messages coming through from the different break-out groups

Next steps

Move from conceptual models to risk analysis and mitigation, to be drafted for next workshop

Consult with stakeholders not represented at meeting.

Agenda Workshop 1

Time	Item	Speaker(s)
08:45	Coffee on arrival	
09:00	Welcome	Cathy Dichmont
	Introduction of participants	Cathy
	Project outline	Rik Buckworth
	Capturing knowledge, insights & values: what we want from the meeting	Rik
	Flow scenarios for the FGARA project	Cuan Petheram/ Geoff Podger
10:30	*Tea break*	
11:00	Building risk assessments – conceptual models to desk top assessment & Workshop 2	Peter B/ Melissa Duggan/Mark Kennard
	Identifying important species	Shane Griffiths
	Conceptual models examples	Rob Kenyon
12:30 - 13:30	*Lunch*	
13:30	Fisheries & ecological values	Break out groups
14:45	Group reports & discussion	Cathy
15:15	Conclusions and summing up	Cathy
15:30	*Close*	

Appendix 2: Workshop 2 report

Assessing the water needs of fisheries and ecological values in the Gulf of Carpentaria

Risk Assessment and Mitigation for Important Species in the Gilbert and Flinders River Systems

Project Workshop #2 Thursday 15 May2014

Rm 3.141 Queensland BioSciences Precinct, St Lucia

Workshop facilitator: Cathy Dichmont (arrives at 9am)

Workshop objectives/expected outcomes

- Provide team members, expert advisors and Reference Group members with update on progress of the project.
- Elicit further information and viewpoints from the workshop group.
- Communication of risk assessment results.
- Discussion of potential mitigation strategies.
- Recommendations for further work.

Introduction

After welcoming the attendees, Rik Buckworth described the objectives for the workshop (as above), and for the project as a whole, indicating that the overall project aim was to deliver a technical assessment of possible impacts to fisheries and ecological values in the Gulf of Carpentaria from key development scenarios specified in the Flinders and Gilbert Agricultural Resource Assessment (FGARA). He briefly described the context of the work and the project methodology. He indicate that, while there would be some input on elicitation of important values, the emphasis for this workshop was to be on risk assessments mitigation and knowledge gaps. He also provided a brief re-cap of the previous workshop.

Shorebirds & Waterbirds

Shorebirds

Peter Rothlisberg presented the draft assessment on the risks to migratory shorebirds and waterbirds from reduced water flows in the Flinders and Gilbert Rivers. He began by explaining the federal Environment Protection and Biodiversity Conservation Act (EPBC) listing on these species and the other migratory bird agreements and international conventions that cover them and Australia is a signatory. He then went on to define the East Asian Australasian flyway and identify wetlands of international importance for migratory birds in the Flyway. The south-eastern Gulf of Carpentaria between the Flinders and Gilbert Rivers is the third most important shorebird site of international importance in Australia. In this region, populations of nine species meet Ramsar criteria for defining wetlands of international importance. There is a hotspot for shorebirds within a 50 km radius around the Normal River. A recent survey (2013) found a 47% reduction in abundance of the nine internationally-important species over 15 years (a feature that is consistent across the flyway). Two of these species have also been listed as Vulnerable to extinction by the IUCN – Eastern Curlew and Great Knot. River flow and flooding provide the nutrients for the benthic in-fauna that these birds feed on. The birds need lots of food as the region is their last stop-over site before long migrations to the northern hemisphere. Studies have found they also return to the same feeding sites each year (irrespective of the amount of food available).

Water birds

There are 62 species of waterbird in Australia and they occur in 9 families. A nationwide survey of freshwater wetlands in 2008 found that the Flinders and Gilbert R catchments in the south-eastern Gulf of Carpentaria had the second largest area of wetland in northern Australia after the Timor region (Kakadu NP). The region also held over 400,000 waterbirds and these was in the five largest populations in Australia. The waterbird that occur in the region are mostly nomadic, moving between wetlands as they dry and fill. In addition, some species are migratory, moving between Australia and nearby countries – Papua New Guinea and Indonesia. Most species nest in the region and this is stimulated by over-bank flooding of rivers. In years of below average wet season rain, breeding is reduced. Thus, reducing the frequency of river flow events that flood off-stream billabongs and wetlands will reduce the frequency of nesting and the number of offspring produced.

Questions

Cuan – Please elaborate on how the likelihood rating was arrived at.

Peter – We assessed the flows and the need for overbank flow. In dry years when overbank flow doesn't occur the breeding success declines dramatically – e.g. Lake Eyre. Estimations elsewhere in the Murray-Darling basin have found that if you reduce flow by a little or a lot the threshold will tip and nesting is reduced.

Bill Sawynok - Lifespan of species – if you don't get breeding events will you get recruitment failure?

Peter – Life spans are variable as most species live > 10 years so variability in flooding events of that interval could overlap with lifespan of the species. Localised extinctions are possible if this occurs but uncertain whether birds from other catchments will re-occupy these areas if flooding subsequently occurs.

Matt Barwick - The shape of hydrographs is also important for fish, two peaks in the flood pattern could be detrimental.

Peter - How the shape of the hydrograph overlaps with the temporal and spatial extent of nesting (+'ve or –'ve) are very important.

Steve Blaber – raised the point that the ecological role of a lot of these species (fish eating birds) are very important for maintaining community structure.

Important species and values for indigenous stakeholders

Kelly Gardner from the Carpentaria Land Council (CLC) provided an overview of the values of freshwater wetland-dependent fauna and flora for indigenous people in the CLC region, including the south-eastern Gulf of Carpentaria. The CLC region covers the area from the NT border to Staaten River including marine areas to northern Gulf of Carpentaria. Kelly listed species important to traditional owners in lower Gulf of Carpentaria. These include turtles, dugong, bluefish/parrotfish, shark, groper, squid, sawfish and a range of other species.

Questions

Matt – On your map with the song and story lines, how do these lines work?

Kelly – There is overlap between areas and some boundaries are not clearly defined, spirits can pass through an area and across boundaries – e.g. dugong blood.

David Milton – Do you think traditional owners would be comfortable with the concept of prioritisation of species under threat?

Kelly – The answer would be no – from experience. Recently, there was a request for prioritisation of cultural heritage sites from QLD – The traditional owners felt all sites are equally important and the whole concept of distinguishing between sites is completely foreign to them! The same would be the case for species – all species are equally important.

Marcus Barber commented that in-country based fishing seasonal shifts in species abundance can be very important to sustain livelihoods and social viability of traditional owner communities. It isn't just the amount of one species but the cumulative seasonal flux and movement of all species that sustains food resources and income.

Important fish species and conceptual models

Rob Kenyon took the workshop through the conceptual models of several of the important fisheries species. He explained the types of habitats each species relied on and the role of river flow and flooding in maintaining that functionality. Conceptual models were developed for important commercial and recreational fisheries species such as banana and tiger prawns, mangrove jack, mud crabs, barramundi and sooty grunter. In addition, conceptual models were also developed for two species of fish of conservation concern. These were Large-toothed Sawfish and Spear-toothed Shark.

Rob then presented each conceptual model and highlighted the predicted effects to each species of reduced flows under the proposed off-river storage removals. These effects were separated into effects on periods of high and low flow and the number of zero-flow days predicted under the assessed scenarios.

Questions

There were no questions of this presentation.

Qualitative risk assessment: ecological values and fisheries

Shane Griffiths presented the results of the qualitative risk assessment of the fisheries and ecologically-important species and species-groups. Shane began by defining the process of the risk assessment and why this approach was chosen. He explained that overall risk was made of the product of two components – likelihood and Consequence. Each component was rated from 1 – 5 and the justification required to score each ranking was explained. Overall risk was then defined according the value of the product of the two ranks. Shane then talked about where the approach had been used previously to assess the threats to fisheries and fishery species with limited ecological and fisheries data. A total of 49 species were assessed with this approach including species important to commercial and recreational fisheries, ecosystem keystone species (defined by an Ecopath model) and species of conservation concern or ecological importance. He then showed the template for the written assessments that would support each species' risk assessment and justify the scores. Finally, Shane went through the process of how the likelihood and consequence scores were developed for a range of examples that were relevant to the audience. This including the predicted effects of reduced flow on that species.

Questions

Bill Sawnock – commented that Golden snapper (*L. johnii*) have become iconic for recreational fishery and increasingly targeted.

Are Narrow-toothed sawfish trunks still able to be taken in offshore gillnet fishery?

Yes. (Note: this was checked during the meeting by Ton Ham, Fishery Manager for the Qld Gulf Fisheries. He indicated that retention of trunks (or any other parts) of any sawfish species is no longer permitted.).

Cuan – Why are flows expressed as mean flow and not as median flow in Rob Kenyon's talk?

Rob – This was to be consistent with the format that flow scenarios were presented in the FGARA report.

Cuan – What specific baseline flow scenario is actually being assessed? Differences between Rob's and Shane's described baseline (Greenhills or Greenhills+Dagworth).

Shane – The risk assessment is not quantitative as we do not have any data on the specific responses of different species to predicted reductions in flow. So species' responses to the actual flow being assessed are only general.

Cuan – Appear to be subtle differences between the responses to Moderate and High risk categories. Is scoring certainty incorporated in risk scores?

Shane – No, this risk assessment approach is qualitative and has no capacity to estimate the uncertainty around each risk categorisation.

Steve Blaber – Which species of mullets are being assessed? – *Liza* spp. Need to make sure it is clear assessments are being assessed against same baseline.

Matt - What effect will changing climate have on the rainfall?

Cathy - Likely to be similar to current climate according to global climate models. Frequency and number of cyclones are difficult to model.

Kelly - Need to include additional species of cultural significance to traditional owners – whiptailed rays, leopard rays, and freshwater lilies, leatherjackets (possibly *Monacanthus spp* or *Scatophagus argus*).

Shane - Can we get a photo of the leatherjacket species (from Kelly)? The leatherjackets are caught by seine nets. (Note: an exchange of photos later confirmed that the species was *S. argus*)

Richard Pillans - Gulf snapping turtle is not in Flinders and Gregory R

Shane – We will look at Kennard's suitability model.

Bill Sawynock – commented that iconic recreational species seem to be covered.

Julie Robins - Mangroves and seagrass are groups that need assessing. Use Boyne R modelling.

Michele Burford – commented that zooplankton communities not being assessed but has been shown in Murray-Darling that regulated flows affect composition and may have caused lack of recruitment by Murray Cod due to loss of key species. Look at Gina Newton's work on freshwater zooplankton for advice. Peter Rothlisberg - Frank Comans' work on pond zooplankton can be relevant. Check Neil Humphrey's work. Julie Robins also commented that nutrients from proposed sugar cane may cause changes in the zooplankton composition. Different situation to that found on east coast. Michele said that estuaries are nutrient poor and productivity driven by flows. Coastal productivity higher due to mixing by tides.

Kelly asked that the assessment bring out species of birds iconic to indigenous communities such as Brolga, magpie geese and coastal birds of prey. The project also needs to assess risk to aquatic plants important to indigenous communities.

Bill Sawynock - Need to take into account secondary impacts on species from habitat impacts. Barra change sex at between 75 – 85cm. Emigration is about 10% in high flow years and nil or negligible in low flow years. Data are lacking of the specific distribution/ abundance of barra in different freshwater/estuarine habitats. Barra are opportunistic in both upstream and downstream movements – if flow is sufficient, then movements will occur. Is timing of flows being considered as well as total flow volume?

David Vance – responded that Banana prawn emigration differs between years and between river systems. He found juvenile banana prawns living in salinity 50 which then moved offshore after the first large rain and none remained. In another system, prawns emigrated continuously throughout wet season. Banana prawn life cycle is more complex than indicated. Emigration depends on timing of flow events. In the Ord River, no bananas occur there under current regulated flows. They require salinity gradient and pulse flood events to maintain environment. Maybe consequences of water extraction can be reduced. Some recruitment to the river will occur unless completely fresh. Risk will be higher on resident juveniles and offshore emigration so risk will stay the same.

Richard Pillans - Adult sawfish only occur offshore. Only juveniles occur in freshwater. Dams will affect river heights in middle reaches and movements upstream. Local extinctions are possible if connectivity is reduced. Significant declines in populations known by indigenous people already. Kelly said that some sawfish were seen above dams after large wet season floods. Traditional owners were re-settled from the middle and upper Flinders and Gregory R catchments (several decades ago). Pastoralists have also already erected illegal impediments to river flow. All sawfish are protected in Qld legislation and not allowed to be taken.

Shane - Mangrove jack are potentially more threatened by developmental trawl fishery than agricultural development.

There was a general discussion about the importance of river flow to maintain seagrasses offshore. River flow may be important to promote increased nutrients for seagrass and changes in the quality and quantity of seagrass has been detected in prawn catch trends. Prawn distribution in Gulf of Carpentaria reflects habitat availability and so overall seagrass extent – little in SE Gulf of Carpentaria. Habitat issue depends on scale of assessment (Gulf of Carpentaria or localised). Seagrass will colonise if flow is low.

Cuan commented in regard to the waterbird risk assessment that hydrographic models show that under all flow scenarios overbank flow frequency will be similar to pre-extraction flows especially in Flinders and less so in Gilbert. DSITIA will finalise models of frequency and extent of overbank flows under all scenarios and these will be available in a couple of weeks. They will also generate average recurrence intervals (ARIs) for modelled scenarios. Proposed dams will not be regularly full and so outflow will be infrequent. However, need to review waterbird likelihood score in light of floodplain inundation modelling.

Meeting agreed shorebird score Consequence = 3 and Likelihood = 4.

Quantitative risk assessment: fisheries (banana prawns and barramundi)

Peter Bayliss explained how he undertook his semi-quantitative assessment of flow reduction on Banana Prawn and Barramundi catch. He said that these species are caught in many catchments apart from Gilbert and Flinders R. Two main data sets were used in the analysis:

121 years of river flow and 42 (not 38) years of commercial banana prawn catch. Both mean and median data were examined. Peter found a similarity between wet year flows for both systems. NPF zones 7, 8 and 9 were used to define the catch from recruits from these rivers, but spatial of the commercial catch was very fine. White banana prawns *P. merguiensis* forms the bulk of the commercial catch, mostly in zones 7 and 8 (SE Gulf of Carpentaria). Peter describe that he used multiple regression analysis where catch = constant+effort+flow (± error).

The model found that most variation in banana prawn catch was due to fishing effort, but that river flow explained 8% of the variation in catch in zones 7/8 10% for zone 9. Banana catch vs. effort and vs. wet year flow were strongly positively correlated.

Monte Carlo simulations were made to assess the impact of water extraction on recreational and commercial barramundi catches. Water lost to end of system flow from FGARA development scenarios were linked to fisheries production. Peter found a10% reduction in banana catch in zones 7/8. He then presented a graph showing % reduction in catch of Banana prawns vs. water harvest. For barramundi, the majority of catch in the region in zone AD18 is from the Flinders River. Barramundi most important catch by weight (30-50%) to the gillnet fisheries in this zone. Barramundi catch was significantly correlated with flow in both systems – stronger in Flinders than Gilbert. Peter estimated a 5-7% reduction in barramundi catch under Extraction scenario 2. He also found a strong decadal signal in trends in flow data as in NT. This decadal cycle in flows fits well with patterns in Magpie Geese count data.

Questions

How does this analysis flow into risk assessment? Depends how you make a living – could add to a lot of money for dependent people. Need to quantify what is an acceptable level of catch reduction. Also need to balance economics of agricultural development vs. social impacts of reduced livelihood for local community. The socio-economic component is a key player. Peter stated that more complex modelling would take another six months work and it would need to account for the Norman R catchment. If more complex modelling were undertaken, it would need to use a range of development scenarios not just FGARA scenarios. But caution is needed - the models may not be the real world and FGARA is not worst case scenario as more development is on the horizon.

Managing for environmental flows

Michele Burford talked about the potential effects of water extraction and damming. These include reducing productivity, loss of refugia, connectivity, excessive nutrient, high sediment and chemicals loads and loss of floodplain habitat and reduction in scale and duration of flooding. Flooding is important because hydrology drives food web structure, overriding local effects. Flooding confers stability and persistence to food web dynamics. Changes in flooding will reduce resilience of rivers. For barramundi, 20% of their food comes from river, 35% from the floodplain and 45% from the estuary. Saltpans become very productive during wet season and algae in the saltpan form the basis of food web. Need 10 days wetting to get algal production going. So in some years there is no productivity from these habitats. Fish may need to utilise this habitat for 2-3 months to feed and grow. How many years are the pans flooded – effects fish production. Hence, critical flow requirements to maintain species life history. Species differ in the relative importance for them of different riverine habitats. For example, we do not know how long refugial pools last in these rivers. Flood pulses are needed for immigration and flood pulse emigration into coastal areas. The proportion of fishes in Norman R that are migratory is very high. Need more off-channel dams than in-channel dams as well as fish passages that are effective. Flow regimes also need to mimic natural conditions. If farms produce excessive nutrients and sediments it will affect maintenance of the riparian zones. Water quality should be maintained according to Australian guidelines. Michele completed her presentation by stating that there were lot of knowledge gaps about effects in these systems.

Questions

Are water quality guidelines any use as they are not designed for that part of the world? The water quality guidelines are ok for chemical pollutants but they do not address nutrients. What about cumulative impacts? Are dams big enough to allow environmental flows? Or how much can they take allowing for environmental flows. Michele said that this was a question of seasonal flow and more exact definitions were required of what water was needed. There followed a general discussion about how to play around with delivery of water and how that is managed and the same for pesticides and water quality. The workshop felt there was more value out of concentrating on management of flows. In order to do this, critical data are missing. Managers need key things to focus on e.g. early season flows for banana prawns. What seasonal flow events are most beneficial? Flow will need to be managed for key species then and many other s will benefit. First flow after the dry season may be most significant.

Values, risks and mitigation strategies, key knowledge needs

Short notes from breakout groups

Fisheries

Mitigation

We followed the top down approach described earlier in the meeting. We addressed the most important species – i.e. those that are important to various sectors and have high risk scores.

Banana Prawns - for emigration (January –May?)

Barramundi - ensure flows December – February+

-ensure connectivity (lateral and longitudinal)

Sawfish -ensure connectivity

Mullets -ensure connectivity

King-threadfins, pikey bream, mangrove jack and several others

-barramundi mitigation would support

Blue threadfin and mudcrabs

-more complex and knowledge gaps mean that barramundi mitigation might not be sufficient for their life histories

Knowledge Gaps

Population dynamics, for many species, is poorly known

Responses to flow at fine scales (spatial, temporal)

Potential socio-economic impacts of losses to the fisheries

Barramundi – critical points of life history at fine scales

Indigenous social impacts of changes in fish abundance

Glyphis sharks

Research and Monitoring Needs

Monitor salinity at the mouth

Economic and social values and impacts of water extraction

Recruitment monitoring of barramundi, comparing Norman R with Flinders and Gilbert in a Before/ After study

Catch and effort data from fisheries to be reported in a finer scale

Sawfish: flow management must not contribute to their decline

Sawfish: development of suitable fish passages

Biodiversity and habitats

Mitigation

Offsets**:** protected or closed areas (e.g. in an adjacent river

Habit restoration elsewhere

Research offsets –targeted to address potentially impacted species/ ecosystems / habitats

Flow (=water height) required to maintain connectivity

Knowledge Gaps

Baseline (eco) community studies / whole of system

Habitat use/ connectivity/ life history (electronic tagging) for key species

Potential for regime shifts (e.g. as in intermittently closed estuaries)

Full trophic linkages / dependencies

What are the stimuli for movements (e.g. flow, salinity, oxygen, food?)

Connectivity and other ecosystem effects

Mitigation

Recommend what you do NOT want, if you are not sure WHAT you do want

Stick to broad statements –can build upon these

First pass dry season flow that permits full connectivity

Spell-length specification applied to floodplain inundation? Conceptual support if no time for specific analysis

Split into longitudinal vs. lateral for key species such as barramundi

Capture important cues for bananas –timing

i) emigration
ii) maintain salinity gradient

No structures to impede floodplain

Need to mitigate:

i) Annual ARI events
ii) 5-10 year ARI events

Seasonal ARI

Knowledge Gaps

Links between flood plume and coastal productivity

Floodplains -dynamics, spatial importance

-source of sink of nutrients

Basic Knowledge of Processes:

Productivity

Sediment and nutrients

Does food availability limit fisheries?

Ecosystem drivers –top down/ bottom up control

Growth and mortality of prawns to identify important flow components

Dry season waterhole refugia

Explore models that explore impacts on fisheries - more complex ones such as Ecomodeller or Ecosim.

Research and Monitoring Needs

Fish and prawn use of floodplains in Wet season –related to stage height

Calibrated 2D salinity model of estuary under various flows

Is there good underlying data – soil type, DEMs, Nitrogen, Phosphorus - for the estuary?

Change in habitat distribution/ mapping

Essential to re-evaluate system understanding and allocation in a system of high risk/knowledge scarcity

Social and economic values

This team focussed o the knowledge gaps and thus research and monitoring requirements

Knowledge Gaps

Indigenous

1. A major issue is that there is no good handle on key aspects of the indigenous fishery in these areas – effort and catch levels, socio-cultural valuations, and/or impacts that changing water flows would have. This knowledge gap has also been identified by local indigenous groups - the Kurtijar, the downstream group in the Gilbert, and the Gkuthaarn and Kukatj, the downstream group in the Flinders. Both groups note it as a key issue in their country-based management plans that are close to finalisation and want further work to be undertaken.
2. Flow-dependent species other than those targeted for fishing effort might be crucial to local people. For example as indicators of key events in the seasonal cycle. There are good seasonal calendars from elsewhere but not our study area.
3. The Kurtijar people own the major cattle station of Delta Downs and are already greatly concerned about the existing effects of saltwater intrusion and general salination. There are concerns these existing processes could be worsened by reduced flow.

General

1) Different subcommunities/subpopulations in the Gulf have different species and/or waterflow dependencies and identifying and mapping these dependencies would be useful, even if placing valuations and quantifications on the dependencies is not possible with the resources available. For example:
    a. impacts on coastal fish might be really important for local residents, particularly indigenous residents whose effort is usually shore-based and with simple gear
    b. impacts on the offshore prawns might have significant regional impacts in Cairns etc., potentially greater than impacts on particular Gulf communities such as Normanton.
    c. The timing of particular impacts may be crucial to particular communities (e.g. the availability of particular resources for religious purposes at indigenous ceremony times, or as nutritional stabilisation during periods of low cash income)
2) Following from 1) further information is required on how the economic and social impacts of water development might vary geographically. E.g. Fishers working in the Gilbert and Flinders may live and have families that are based in Cairns. Others that use the area may just be passing through (e.g. tourists, Grey nomads) and be relatively unaffected provided basic services are available.
3) Following from 2), the institutional and organisational structure of any development, and potentially the identity of the developers, is crucial to where the impacts and benefits of resource development might fall. Will financial benefits accrue outside the Gulf and/or overseas? Further market and non-market valuation processes are needed to assess and ultimately quantify the costs and benefits of water development for downstream users
4) The lack of knowledge about econo-social issues constrains the identification and evaluation of mitigation options and the policies and processes required to make such mitigation measures effective.
5) The cumulative social and economic impacts of multiple developments on social and economic systems needs further understanding, both multiple agricultural developments and combinations of developments (e.g. Agriculture and mining).

Research and Monitoring Needs

Studies in the downstream section of the Flinders R and Gilbert R to understand and monitor specific flow-dependent indigenous assets and the ongoing level of indigenous fishing and hunting effort that might be impacted by water development

Mapping of resource dependencies is needed to provide likelihoods of different sorts of impacts, including valuations of dependencies and the ways that they might constrain mitigation options.

Timeframe

It was noted that the Water Resources Plan puts a real time constraint on actions, in that once in place it might be some years before it is reviewed

Agenda

Time	Item	Speaker(s)
08:00	Coffee on arrival	
08:30	Welcome	Peter B
	Introduction of participants	Round table
	What we what we want from the meeting	Rik
	Recap of workshop #1	Rik
09:00	Shorebirds & Waterbirds	Dave M/Rotho
09:15	Important species and values for indigenous stakeholders	Kelly
09:30	Important species and conceptual models	Rob
10:00	*Morning tea*	
10:15	Qualitative risk assessment: ecological values and fisheries	Shane/Rob
11:15	Have we captured the important species/groups & values?	Julie & group
	What species/risks have not been identified, any changes?	discussion
12:00– 12:30	*Lunch*	
12:30	Quantitative risk assessment: fisheries (prawns & barra)	Peter B
13:00	Managing for environmental flows	Michele
13:30	Values, risks and mitigation strategies, key knowledge needs	Break out groups
	- How can we mitigate risks? - Where are the knowledge gaps? - What are the research & monitoring needs?	
15:30	*Afternoon tea*	
15:45	Group reports & discussion	Cathy
16:45	Conclusions and summing up	Cathy
17:00	*Close*	

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