EVALUATION OF THE EASTERN TORRES STRAIT REEF LINE FISHERY

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Non-Technical Summary

T1.1 Evaluation of the Eastern Torres Strait Reef Line Fishery

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**Objectives**

1. Analyse historical catch and effort data of non-indigenous fishers and Torres Strait Islanders to determine spatial and temporal patterns in the eastern Torres Strait (ETS) commercial reef line fishery.

2. Develop an on-going sampling program to assess spatial and temporal patterns in catch and catch composition of the ETS commercial reef line fishery.

3. Document the catch composition of non-indigenous fishers and Torres Strait Islanders in the ETS commercial reef line fishery.

4. Collect and analyse samples of key reef fish species within the ETS commercial reef line fishery, including leopard coral trout (*Plectropomus areolatus*), Islander trout (*P. maculatus*) and common coral trout (*P. leopardus*) to estimate fundamental biological parameters required for the assessment of the ecological sustainability of their harvest.

5. Extend and apply to the ETS, management strategy evaluation (MSE) models from the reef line fishery of the Great Barrier Reef World Heritage Area (GBRWHA) to evaluate alternative potential management strategies for the Torres Strait.

During the course of the project additional research priorities were identified at Torres Strait Finfish Working Group meetings. A complementary project (T1.14) was developed in response to these research priorities. Two additional objectives from T1.14 are addressed in this report:

6. Collate Torres Strait Islander commercial catch records from Council freezers on Badu, Poruma (Coconut), Iama (Yam) and Waibene (Thursday) Islands.

7. Collect and analyse barramundi cod biological data from the ETS reef line fishery.
Summary

Torres Strait Islanders have harvested marine resources for subsistence and cultural reasons for centuries, but today commercial fishing is the most economically important industry in the Torres Strait. There are numerous commercial fisheries currently operating in Torres Strait including prawn, tropical rock lobster, Spanish mackerel, barramundi, pearl shell, crab, trochus, beche de mer and reef fish. For most of these fisheries, however, ecologically sustainable harvest rates are not known and the information upon which these rates are estimated is limited. This report is focussed on providing information to assist the sustainable management of the line fishery for coral reef fish in the eastern Torres Strait (ETS).

Reef fish are predominantly caught by commercial fishers in Torres Strait, although relatively minor catches are reported from traditional subsistence fishers and recreational and charter fishers. Commercial harvest of reef fish mainly occurs in the ETS and is shared between Torres Strait Islanders and non-indigenous fishers. Concerns have been expressed by all stakeholder groups over the lack of documented information about the status and exploitation of the ETS reef line fishery. These concerns are primarily driven by questions about the sustainability of current harvest rates of key reef fish species within the Torres Strait and perceived risks of over-exploitation of finfish resources. Exacerbating these concerns is the fact that no formal assessment of the fishery has been done and no resource allocation or management strategies directed at the special circumstances in the Torres Strait currently exist.

Management arrangements for the ETS reef line fishery are now under review, with new management arrangements likely to regulate commercial effort in the fishery explicitly through a total allowable commercial catch. However, limited historical information about the fishery or its main target species presents significant problems for the development of appropriate management strategies for the fishery. In this report, we provide a detailed description of the harvest patterns and effort dynamics of the Islander commercial and non-indigenous commercial sectors. We describe the population biology of the key target species of the fishery: common coral trout (*Plectropomus leopardus*), bar-cheeked coral trout (*P. maculatus*), passionfruit coral trout (*P. areolatus*) and barramundi cod (*Cromileptes altivelis*). Finally, we evaluate the likelihood for alternative combinations of management strategies for the ETS reef line fishery (RLF) to realise the objectives of fishery stakeholders.

Analysis of a range of data sources including commercial logbooks, community freezer records, voluntary logbooks and observer surveys demonstrated that bycatch is a significant component of the catch for both Islander and non-indigenous commercial sectors of the ETS RLF. Substantial differences in the harvest patterns and effort dynamics were also identified between the sectors. Differences between sectors were observed in species composition and spatial and temporal patterns in catch, effort and catch per unit effort. These results highlight the inherent variation in the catch and effort dynamics between the two commercial sectors of the ETS RLF and provide valuable information for the development of future assessments and appropriate management strategies for the fishery. More reliable estimates of harvest patterns and effort dynamics for both sectors obtained from observer surveys will also assist in resolving issues relating to allocation of reef fish resources in Torres Strait.

The spatial distribution and population biology differed significantly among coral trout species in the ETS. In particular, the population biology of *P. areolatus* differed substantially to that of *P. leopardus* and *P. maculatus*. The length-weight relationship,
length and age distributions, growth, peak spawning season and length and age at sex change of *P. areolatus* differed significantly to *P. leopardus* and *P. maculatus*. Age distributions and size at sex change were the only two parameters that differed significantly between *P. leopardus* and *P. maculatus*. These differences suggest that *P. areolatus* may respond differently to fishing pressure than *P. leopardus* and *P. maculatus* and, therefore, may require separate management arrangements.

This study provides some of the only estimates of biological parameters for *C. altivelis* and the first estimates for populations in Torres Strait. Populations of *C. altivelis* in Torres Strait were found to exhibit a moderate longevity, with a relatively fast growth rate in the first few years of life, followed by a period of relatively slow growth. *C. altivelis* were also confirmed to be protogynous hermaphrodites. Generally, the estimates of biological parameters for *C. altivelis* from Torres Strait were similar to those estimated for populations on the GBR. Therefore, given that the two regions are adjacent, combining the information from both studies provides valuable information for the management of *C. altivelis* populations throughout the GBR and Torres Strait.

This project also provided a formal context within which to evaluate a range of alternative management strategies, related to the harvest and conservation of coral trout, the major target species of the ETS RLF. Specific operational management objectives, performance measures and management strategies were defined for coral trout in consultation with stakeholders and formally evaluated using the Effects of Line Fishing Simulator (ELFSim). Results of simulations indicate that a seasonal closure was a good measure for addressing conservation, stock and economic objectives. Closing areas to fishing and increasing the minimum legal size were generally good strategies for stock objectives, but performed poorly for harvest and economic objectives. Lowering effort was the more robust strategy for achieving stock and economic objectives. Trade-offs among conflicting management objectives were highlighted by the effect that although reducing effort increases the chances of achieving these objectives, it reduces the likelihood of achieving the harvest objective.

As a result of this research, we have highlighted the consequences of different options for managing the ETS RLF. These have been put forth in a manner that is comparative rather than prescriptive, stressing the trade-offs among the many diverse objectives held for the fishery. Such a process aids in decision-making in a frank and transparent manner.

**Keywords**

Torres Strait, Queensland, reef line fishery, coral trout, *Plectropomus*, Barramundi cod, *Cromileptes altivelis*, reef fish, catch, effort, fisheries management, management strategy evaluation
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This project was funded by the Cooperative Research Centre for the Great Barrier Reef World Heritage Area (CRC Reef Research Centre Torres Strait Program), and James Cook University.
Background and Need

The project arose in response to major concerns expressed by stakeholder groups over the lack of documented information about the current status and exploitation of the eastern Torres Strait (ETS) reef line fishery (RLF). These concerns were primarily driven by questions about the sustainability of current harvest rates of key reef fish species within the Torres Strait and perceived risks of over-exploitation of finfish resources.

Commercial harvest of reef fish in the ETS is shared between Torres Strait Islanders and non-indigenous fishers, but to date no formal assessment of the fishery has been done, and no allocation or management strategies directed at the special circumstances in the Torres Strait exist. Additional concerns include the considerable latent effort that exists in the commercial sectors, potential for increased fishing pressure driven by expansion of the live fish trade into the Torres Strait, and future development of charter boat and tourism operations in the ETS.

Whilst in some respects the reef line fishery in the ETS has much in common with the east coast reef line fishery of the Great Barrier Reef World Heritage Area (GBRWHA), there are also important differences. In particular, the needs of the Torres Strait Islanders have no parallel in the east coast fishery and the species composition of commercial catches in the Torres Strait likely differs from that of the east coast. These differences mean that assessments of management strategies for the Torres Strait require dedicated research to supplement information from research on the east coast.

Accordingly, there is an urgent need for research into the catch and effort characteristics of the ETS RLF, the biological characteristics of key target species, and the suitability of alternative management strategies for the specific circumstances of the Torres Strait. The proposed research provides the necessary pre-requisites to future application to the Torres Strait of formal management strategy evaluation (MSE) tools developed for the GBRWHA reef line fishery, and will provide the initial application of these assessment tools. Moreover, recent conflict between the stakeholder groups emphasises the need for research into the ETS RLF to resolve ever-increasing resource allocation and sustainable utilisation issues.
Benefits

This project has provided direct benefits to the assessment and management of the ETS RLF. Formal assessment of historical catch and effort data has enabled stakeholders, AFMA, the Torres Strait Regional Authority (TSRA), the Torres Strait Fisheries Management Advisory Committee (TSFMAC), and the Torres Strait Fisheries Scientific Advisory Committee (TSFSAC) to approach future management decisions from a more informed basis about past and present fishing practices. Documentation of catch composition has clarified the degree of overlap between commercial and traditional harvest, thereby assisting in future decisions about resource allocation and appropriate regulatory frameworks for the commercial fishery. Thorough documentation of by-catch/by-product from the fishery has addressed key ecological sustainable development issues relevant to the Environment Protection & Biodiversity Conservation Act and Native Species Export regulations. Estimation of fundamental biological parameters of key harvest reef fish species has been paramount to the sustainable development of the fishery, particularly as there was little or no information known about two of these species (*Plectropomus areolatus* and *P. maculatus*). Outputs from the project also built on previous research of the CRC Reef Effects of Line Fishing (ELF) Project (Mapstone et al. 2004), the AFMA funded review of the ETS reef line fishery (Mapstone et al. 2003), and the CRC Torres Strait review of Islander commercial catch history (Begg and Murchie 2004). All stakeholders and managers involved in the ETS reef line fishery have benefited from this project through the provision of information for improved management of harvested stocks in the medium-long term. Active engagement of stakeholders in the research process and extensive liaison during the research has resulted in improved understanding of the research, assessment and management process.
1. Introduction

The harvest of marine resources is fundamental to the livelihood of Torres Strait Islanders for reasons of tradition, culture, economics and subsistence. Although Torres Strait Islanders have harvested marine resources for subsistence and cultural reasons for centuries (Johannes and MacFarlane 1991), today commercial fishing is the most economically important industry in the Torres Strait and provides the greatest opportunity for financial independence of Islander communities (TSPZJA 2003). Islanders also share many of the marine resources in Torres Strait with non-indigenous commercial fishers, which has led to conflict over allocation of these resources. There are numerous commercial fisheries currently operating in Torres Strait including prawn, tropical rock lobster, Spanish mackerel, barramundi, pearl shell, crab, trochus, beche de mer and reef fish (TSPZJA 2003). For most of these fisheries, however, ecologically sustainable harvest rates are not known and the information upon which these rates are estimated is limited.

The focus of this research is on the reef line fishery (Torres Strait finfish fishery) that operates around the eastern Islands of Torres Strait (Fig. 1.1). Other Torres Strait fisheries, including the line fishery for Spanish mackerel, are not discussed.

![Fig. 1.1. Map of Torres Strait indicating the area in which the eastern Torres Strait reef line fishery operates. Commercial fishing is not permitted west of 142°31'49" and is restricted to within the Australian Jurisdiction, Australian outside but near area and Australian Territorial Seas north of the Seabed Jurisdiction Line.](image)

1.1. The Torres Strait Reef Line Fishery

The reef line fishery in Torres Strait is comprised of five sectors: 1) Islander fishers taking fish for subsistence and operating under their own methods from island communities, 2) Islander fishers taking fish for commercial sale, also operating from
island communities, 3) Non-indigenous commercial fishers operating from large, mobile fishing vessels, 4) Non-indigenous recreational fishers, and 5) Tour or charter operators hosting recreational fishing trips. The latter two sectors are relatively minor components over most of Torres Strait, occurring in the southern areas near Thursday Island and Cape York (Mapstone et al. 2003). All sectors have access to the same grounds in the eastern and central Torres Strait, though commercial line fishing by Islanders and non-indigenous fishers is not permitted west of 142°31’49” (Fig 1.1).

A large number of demersal reef fish species are captured in the Torres Strait reef line fishery (Mapstone et al. 1996, 2003). Most sectors target or take the same suite of species using similar simple hook and line gears, although the Islander fishers traditionally use a diversity of gears including spear, hook and line, stone walled traps (sai) and poison to target some species not targeted frequently by the other sectors (Fuari 1991, Johannes and MacFarlane 1991, Poiner and Harris 1991). Coral trout (Plectropomus spp.), barramundi cod (Cromileptes altivelis), emperors (Lethrinus spp.), tropical snappers (Lutjanus spp.) and a variety of cods (Epinephelus spp.) are the main targeted species of the reef line fishery in Torres Strait for both Islanders and non-indigenous fishers (Fig. 1.2) (Poiner and Harris 1991, Mapstone et al. 1996, 2003, Begg and Murchie 2004).

This research is focused on the two commercial sectors (Islander and non-indigenous) in the eastern Torres Strait, in areas where non-indigenous commercial line fishing is allowed (Fig. 1.1) and where concerns about the status of the fishery are greatest. The research does not cover the Islander subsistence fishery, which is covered in detail in the related post-graduate thesis of Ms Sara Busilacchi, or the relatively minor recreational and charter fisheries.

1.2. Commercial Islander fishing

Commercial fishing for reef fish by Islander fishers began in the late 1980’s (Mapstone et al. 2003, Begg and Murchie 2004) after processing facilities and large freezers were established on some islands. The three main islands in Torres Strait involved in the fishery are Murray (Mer), Darnley (Erub) and Yorke (Masig) Islands, although small quantities of reef fish may be taken from other islands. Reef fish are
harvested commercially by Islander fishers using small (4-6 m), outboard powered aluminium or fibreglass boats (dinghies) which allow Islanders to fish reefs beyond their home island (Fig. 1.3) (Mapstone et al. 2003, Begg and Murchie 2004).

Most fishing trips are completed in one day but occasionally span two or more days with fishers staying on other islands or cays. There is often more than one fisher per dinghy, although the number of Islander fishers in each dinghy varies greatly. Fishing is typically by 50-130 lb breaking strain handlines, usually with one line per fisher and one hook, sinker and bait per line. Baits used include pilchards (*Sardinops pilchardus*) or local sardines caught from island beaches using cast nets. Fishers typically leave the island in the morning, fish a number of locations at anchor (hangs) and return to the island in the afternoon.

Those species that are harvested commercially are typically sold to the community or council freezer of the fisher’s home island, while those that are harvested, but not sold, are kept for subsistence (Johannes and MacFarlane 1991, Mapstone et al. 2003). Fish are processed as either gilled and gutted or filleted product and then frozen at the freezer. Product is then shipped weekly to mainland Australia (Cairns) via barges.

A recent review of Islander commercial harvest and effort records indicated that annual harvest of reef fish has increased substantially since commercial Islander fishing activities began (Begg and Murchie 2004). Annual harvest of reef fish from Murray, Darnley and Yorke Islands reached almost 29 t in 2002 from 1064 days of fishing effort and 111 individual fishers; albeit given significant issues with data completeness and reliability (Begg and Murchie 2004).

*Fig. 1.3. Islander fishers preparing for a day’s fishing from Murray Island.*
1.3. Commercial non-indigenous fishing

A commercial fishery for reef fish by non-indigenous fishers has existed in Torres Strait since the 1950’s (Haysom 2001). Commercial fishing by non-indigenous fishers was previously done from vessels travelling to Torres Strait from ports on the east coast of Queensland. More recently though, non-indigenous commercial fishing operations have been able to remain in Torres Strait almost year round (Mapstone et al. 2003) by offloading product and replenishing supplies from barges that service Torres Strait from ports on the east coast (Begg and Murchie, 2004).

Non-indigenous commercial fishing operations for reef fish are similar to the line fishing operations on the Great Barrier Reef (GBR), whereby fishing is done from a number of outboard powered aluminium or fibreglass dories (typically 4-6 m long), which are tendered to a larger (8-20 m) primary vessel (Fig 1.4) (Mapstone et al. 1996, Scott and Mulrennan 2003). Similar to Islander fishers, fishing is by 50-130 lb breaking strain handlines, usually with one line per fisher and one hook, sinker and bait per line. Unlike Islander fishers, however, there is rarely more than one fisher in each dory. Both pilchards (Sardinops pilchardus) and, less frequently, locally caught fin fish (hard bait) are used for bait.

Fishing generally occurs in two ‘sessions’ (AM and PM) per day, with dories moving among several hangs on a reef or shoal throughout the session. Dories work independently at varying distances from the primary vessel during each session. Fish captured in each session are transferred to the primary vessels at the completion of
the session, where they are weighed and processed. To date, nearly all commercial catch has been killed and marketed domestically and internationally either as frozen fillets or frozen gilled and gutted fish. Sale of fish alive for export has been discouraged in the Torres Strait and is now illegal, despite considerable growth in the live reef food fish trade from the reef line fishery along the Queensland east coast (Mapstone et al. 1996, 2001). A recent review of commercial non-indigenous compulsory logbook data (1989-2000) indicated that harvest and effort reached a peak in 1998 of over 160 t of reef fish from over 4000 line days, respectively (Mapstone et al. 2003).

1.4. Management of the Torres Strait Reef Line Fishery

Management arrangements for the fishery are complex as the fishery operates substantially within the Torres Strait Protected Zone and has been subject to direct or indirect regulation by both State and Commonwealth Acts. Prior to April 1 1999, direct fisheries regulation was under the Queensland Fisheries Act 1994 as administered until June 30 1999 by the Queensland Fisheries Management Authority (QFMA) and since July 1 1999 by the Queensland Fisheries Service (QFS). Since April 1, 1999, the fishery has been managed jointly by the Torres Strait Protected Zone Joint Authority (PZJA) under the Torres Strait Fisheries Act 1984. Administrative support for the PZJA is provided by the Australian Fisheries Management Authority (AFMA) and the Queensland Department of Primary Industries and Fisheries (QDPI&F, formerly QFS and QFMA).

The fisheries management measures for the fishery currently include a cap on the total number of commercial licences, recreational bag limits, minimum size limits for a range of species, gear restrictions and a moratorium on the sale of live fish. These regulations apply to all sectors of the fishery with the exception that Islander peoples are not constrained by minimum legal size or bag limits for fish that they take for subsistence, although minimum size limits apply to all fish they wish to sell.

1.5. Objectives

In recent years, major concerns have been expressed by Islanders, non-indigenous fishers and management agencies over the lack of documented information about the current status and exploitation of the reef line fishery in the ETS. These concerns are primarily driven by questions about the sustainability of current harvest rates of key reef fish species and perceived risks of their over-exploitation in some areas. Existing information about the fishery and its associated targeted fish stocks is also limited and patchily distributed (Mapstone et al. 2003).

Despite the significance of the reef fish resource to Torres Strait Islanders and non-indigenous fishers, no formal assessment of the reef line fishery has been undertaken and no resource allocation or management strategies are in place that address the special circumstances that exist in the Torres Strait. In a recent review of the status of the reef line fishery in the ETS, Mapstone et al. (2003) made a number of recommendations for future research important for the management of the fishery. Those recommendations most relevant to this report include:

i) Collation and analysis of all existing information. This includes data from both the Islander and non-indigenous harvest which should be formally analysed for assessments of changes in catch, effort, catch rates and sizes of fish being landed.

ii) Biological research on the species most targeted by the reef line fishery. This includes documenting the species composition and distribution of catches for Islander and non-indigenous fishers, focussing biological research on species for
which little information currently exists (e.g. *Plectropomus areolatus* and *P. maculatus*), and verifying whether biological research done elsewhere can be applied to Torres Strait.

iii) Formal assessments of the likely strengths and weaknesses of potential management strategies for the fishery. Management Strategy Evaluations (MSE) have been developed for the Queensland east coast reef line fishery and could be applied to the Torres Strait with relatively little further development. These MSE tools provide stakeholders with assessments of how likely different management options are to meet specific objectives for the fishery, thereby aiding the choice of the most appropriate medium to long-term mix of management strategies.

Addressing these research recommendations will enable stakeholders to approach future management decisions from a more informed basis, particularly those concerning resource allocation and sustainable harvesting strategies. The objectives of this project, therefore, included the following:

1. Analyse historical catch and effort data of non-indigenous fishers and Torres Strait Islanders to determine spatial and temporal patterns in the eastern Torres Strait (ETS) commercial reef line fishery.
2. Develop an on-going sampling program to assess spatial and temporal patterns in catch and catch composition of the ETS commercial reef line fishery.
3. Document the catch composition of non-indigenous fishers and Torres Strait Islanders in the ETS commercial reef line fishery.
4. Collect and analyse samples of key reef fish species within the ETS commercial reef line fishery, including leopard coral trout (*Plectropomus areolatus*), Islander trout (*P. maculatus*) and common coral trout (*P. leopardus*) to estimate fundamental biological parameters required for the assessment of the ecological sustainability of their harvest.
5. Extend and apply to the ETS, management strategy evaluation (MSE) models from the reef line fishery of the Great Barrier Reef World Heritage Area (GBRWHA) to evaluate alternative potential management strategies for the Torres Strait.

During the course of the project additional research priorities for the ETS reef line fishery were identified at Torres Strait Finfish Working Group meetings. These priorities arose out of concerns about the relevance to Torres Strait of the recent declaration of barramundi cod (*Cromileptes altivelis*) as a no-take species on the Queensland east coast, and that reef fish harvests from islands other than Murray, Darnley and Yorke Islands were not included in estimates of overall harvest for the Islander commercial sector. A complementary project (T1.14) was developed in response to these research priorities. Two additional objectives from T1.14 are addressed in this report:

6. Collate Torres Strait Islander commercial catch records from Council freezers on Badu, Poruma (Coconut), Iama (Yam) and Waibene (Thursday) Islands.
7. Collect and analyse barramundi cod biological data from the ETS reef line fishery.
2. General Methods

2.1. Study Area

This study is focussed on the two commercial sectors of the reef line fishery (RLF) that operate in the eastern Torres Strait (ETS) (Fig. 2.1). The three main islands in Torres Strait involved in the Islander commercial sector are Murray (Mer), Darnley (Erub) and Yorke (Masig) Islands (Fig. 2.1), although small quantities of reef fish may be taken from other islands. The two commercial sectors are restricted in the area in which they can fish, with commercial fishing prohibited west of 142°31’49” (see Fig 1.1).

![Map of Torres Strait](image)

Fig. 2.1. Map of Torres Strait indicating the location of the main islands involved in the Islander commercial sector of the ETS reef line fishery (Murray, Darnley and Yorke Islands).

2.2. Data Sources

Numerous data sources were used to address the objectives of this project including non-indigenous compulsory logbooks, Island community freezer records and research logbooks at community freezers. Observer surveys on Islander and non-indigenous commercial vessels and regular catch samples from both sectors provided additional catch and effort data and biological samples of key species.

2.2.1. Non-indigenous compulsory logbooks

Compulsory logbook programs for non-indigenous commercial fishers operating in Torres Strait were introduced in 1988 by the AFMA and in 1989 by the QDPI&F. Non-indigenous commercial fishers were required to report in one of these concurrent logbooks, although occasionally data were recorded in both. In 2003, a general line fishing logbook for all finfish harvested in Torres Strait was introduced by AFMA, and reporting in QDPI&F logbooks ceased. Accordingly, we obtained catch and effort data from the QDPI&F for 1988 to 2003 and from the AFMA for 1989 to 2004. Data
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were reported at the scale of 30’ x 30’ grids and, since 1993, optionally at the scale of 6’ x 6’ sites within these grids. Fishers also are expected to report the numbers of crew, lines and dories used each day, though these data are optional.

2.2.2. Island community freezer records
Commercial Islander fishing data have been recorded at community freezers on a number of inhabited islands in Torres Strait since 1988. Data reported in freezer records includes the total catch of species or species group for each dinghy day. The number of fishers that fished in each dinghy or the number of lines used is not recorded. Location of catch is only recorded at a very coarse spatial scale and is often not recorded at all.

A number of islands in Torres Strait were visited to obtain catch and effort data from freezer records. The first trip in April and May 2003 was to Darnley, Murray and Yorke Islands, which were known to be the main islands involved in commercial reef line fishing (Mapstone et al. 2003, Begg and Murchie 2004). To determine whether reef fish were commercially harvested at other islands, a second trip in May 2006 was made to Warraber, Coconut, and Thursday Islands. Freezer records were also requested from Badu and Yam islands, which were not able to be visited at the time. There were no records of fishing activities available from Badu, Yam or Thursday Islands and no records of reef fish being caught from Warraber or Coconut islands. Therefore, analysis of Islander commercial fishing for reef fish was restricted to Murray, Darnley and Yorke Islands.

Community freezers commenced operation in different years on each island. As such, data were available for Yorke Island from 1988-2005, Murray Island from 1994-2005 and Darnley Island from 2001-2005.

2.2.3. Research logbooks
Voluntary research logbooks (Appendix 1) were developed to obtain more detailed information on the catch composition of Islander commercial fishers, as catch is typically recorded as species groups in freezer records, including the main target species, coral trout (*Plectropomus spp.*). Logbooks were placed at the freezers on Murray and Darnley islands and freezer staff were asked to voluntarily record the number of individuals of each species sold to the freezer by each fisher on each day. Research logbooks were initially trialled at Yorke Island, but at the time no reef fish were being sold to the freezer. Data recorded in research logbooks between March 2004 and December 2005 were made available from Murray and Darnley Islands.

2.2.4. Observer surveys
Research observers were placed on commercial Islander and non-indigenous fishing vessels to collect detailed catch and effort data that were not available from any existing logbooks (see Appendix 2). Five two-week observer surveys were conducted with the non-indigenous commercial fishers in 2004 (for a total of 52 observer days), while three observer surveys of varying length were conducted with the Islander commercial fishers in 2005 and 2006 (for a total of 10 observer days).

Observers on non-indigenous commercial vessels spent each fishing day in one of the fishing dories. Each dory was sampled until all dories had been sampled and then the order repeated so that no one dory was sampled on consecutive days. Observers remained in the dory throughout each fishing session during the day (usually an AM and a PM session), recording for each hang (specific fishing location); i) time taken to search for the hang (search time), ii) time taken to set the anchor and position the dory on a hang (set time), iii) coordinate of the hang location, iv) depth of
water at each hang, v) time fishing started and ended (fishing time) at each hang, and vi) time, species, length and fate of each fish caught. On return to the primary vessel, the catch weight and species composition were recorded for each of the other dories.

Observer surveys with Islander commercial fishers were similar to the surveys with the non-indigenous fishers with some minor differences. Three 5-8 day trips were made to Murray Island in Torres Strait between May 2005 and April 2006. Unfortunately, observer surveys with Islander fishers were not possible on any other islands due to the reluctance of some communities to participate and very low levels of fishing activity. Observers would accompany commercial Islander fishers opportunistically when they departed the island, as Islander fishers did not fish every day and did not always have adequate space for an observer on every trip. On each survey, an observer would accompany Islanders in their dory to observe the entire day’s fishing activities, which was generally an all day fishing session. Data from a total of 10 observer survey fishing days were recorded for Islander fishers. The same hang and catch details were recorded for the Islander fishers as for the non-indigenous fishers.

2.2.5. Catch samples

Biological samples of each of the coral trout species (P. leopardus, P. maculatus, P. areolatus and P. laevis) and barramundi cod (Cromileptes altivelis) were collected during observer surveys on board non-indigenous commercial vessels (see Section 2.2.4). Additional biological samples of P. leopardus, P. maculatus and P. areolatus were collected in the form of frozen fish frames (head, skeleton and viscera) and whole C. altivelis were purchased from non-indigenous commercial fishers from March 2004 to November 2005. Where possible, the fork length (FL) and whole wet weight (W) were measured and sagittal otoliths and gonads removed from each fish. These samples were used to estimate a range of biological parameters for each species including age, growth, mortality, spawning season, sex ratio and size and age at sex change.
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3. Harvest patterns and effort dynamics of Islander and non-indigenous commercial sectors of the eastern Torres Strait reef line fishery

3.1. Introduction

Catch and effort data have long formed the basis of most stock assessments of exploited fish populations where catch per unit effort (CPUE) is used as an indicator of relative abundance (Beverton and Holt 1957, Hilborn and Walters 1992). Problems associated with the use of CPUE as an indicator of stock abundance have been well documented (e.g. Beverton and Holt 1957, Paloheimo and Dickie 1964, Harley et al. 2001) and are typically associated with a lack of fundamental information about the stock and behaviour of fishers. Commercial fishing logbooks are often the primary source of catch and effort data for fisheries assessments, but in many cases the information recorded in logbooks is reported at broad spatial and temporal scales and, in multi-species fisheries, catch is often reported in species groups (Mapstone et al. 1996). Such data can be insufficient to obtain reliable estimates of catch and effort and may lead to serious biases in estimates of CPUE (Walters 2003). Gaining an understanding of the catch and effort dynamics of a fishery at spatial and temporal scales relevant to the stock and fishery is a fundamental step towards reliable stock assessments and a sustainable fishery.

Commercial harvest of demersal reef fish in the ETS RLF is shared between Islander and non-indigenous fishers (Begg and Murchie 2004). These two commercial sectors both share the same resource, target the same suite of species and operate using similar hook-and-line fishing gears and methods. Despite the apparent similarities between the two sectors, little is known about the harvest patterns and effort dynamics of each sector. Until recently, existing data about the RLF in Torres Strait were scarce and limited to catch and effort information on a coarse spatial and temporal scale from compulsory commercial logbook data for the non-indigenous sector (Mapstone et al. 2003). A recent review of the Islander catch reported in island community freezer records (Begg and Murchie 2004) provided some insight into the catch and effort dynamics of the Islander sector of the ETS RLF. However, the spatial and temporal resolution of existing data from both sectors was too coarse to obtain accurate estimates of catch and effort for the fishery, even for the target species (Mapstone et al. 2003, Begg and Murchie 2004). The lack of sufficient resolution in the data has precluded the development of a formal stock assessment for the ETS RLF which has led to uncertainty about the current status of reef fish stocks in Torres Strait and the sustainability of the fishery.

Detailed information on the resource use for both the Islander and non-indigenous commercial sectors of the ETS RLF is needed to develop an initial stock assessment for the fishery and to assist in sustainable management of the fishery. In particular, sector-specific details on the spatial and temporal harvest patterns, harvested species composition, bycatch composition and distribution and level of effort are needed. This information will be particularly useful given that the current management arrangements are likely to change imminently, with a proposal for a total allowable catch to be shared between both commercial sectors (AAP 2006). Such information will also be useful in achieving federal legislative requirements for sustainable use and management of Australian fisheries (Fletcher 2003).

In this study, we characterise the harvest patterns and effort dynamics of the Islander and non-indigenous commercial sectors of the RLF in the ETS using a range of data
sources including commercial logbooks, community freezer records, voluntary logbooks and observer surveys. In doing so, we provide more reliable estimates of catch, effort and CPUE for the fishery, identify differences between sectors and provide information that is useful for resolving issues related to allocation of reef fish resources in Torres Strait.

3.2. Methods

The research was done over four years (2003-2006) in the ETS (east of 142°30'19"E) (see Fig. 2.1). Several data sources were used including i) non-indigenous commercial logbooks, ii) Island community freezer purchase records, iii) voluntary Islander catch composition research logbooks, and iv) observer surveys on Islander and non-indigenous commercial vessels. Methods for how data from each of these sources were collected are outlined in Chapter 2.

3.2.1. Non-indigenous Commercial Logbook Data

Non-indigenous commercial fishers have been required by QDPI&F or AFMA since 1988 to report their fishing location and catch (by species or species groups) for each day in compulsory logbooks (see Chapter 2). Logbook data from the QDPI&F for 1988 to 2003 and from the AFMA for 1989 to 2004 were made available for this study. These datasets were initially standardised to allow the two datasets to be combined and remove duplicate records. All catch and effort records were then standardised and assigned to reefs (see Section 3.2.6) to allow spatial and temporal estimates of catch, effort and CPUE for the non-indigenous commercial sector.

3.2.2. Island Freezer Records

Data for Islander commercial catch and effort are recorded at the community freezers on a number of islands in Torres Strait (see Chapter 2). For this study, records were available from Murray (1994-2005), Darnley (2001-2005) and Yorke (1998-2005) islands, which are considered to be the main islands involved in commercial reef line fishing in Torres Strait (Begg and Murchie 2004). Catch and effort data from these islands were standardised (see Section 3.2.6) to provide valid estimates of catch, effort and CPUE for the Islander commercial sector.

3.2.3. Research Logbooks

Research logbooks were placed at the island community freezers on Murray and Darnley Islands to record species level catch composition information for the commercially harvested catch (see Chapter 2). Staff from the community freezers voluntarily filled out the logbooks by recording the number of each commercially saleable species from the day’s catch of each commercial Islander fisher. Data recorded in research logbooks between March 2004 and December 2005 were made available from Murray and Darnley Islands.

3.2.4. Observer Surveys

Observer surveys were conducted with both Islander and non-indigenous commercial fishing operations to provide contemporary data for the estimation of the harvest patterns and effort dynamics of the two sectors (see Chapter 2). Observers were placed onboard commercial vessels to provide data from direct observation of fishing activities. Non-indigenous commercial fishing activities were observed for a total of 52 days from five surveys in 2004. Islander commercial fishing activities were observed for a total of 10 fishing days from three surveys in 2005 and 2006.
Detailed catch and effort information was recorded at the dory level for each day of fishing, which was used to obtain accurate estimates of catch, effort and CPUE for the two commercial sectors.

3.2.5. Catch Composition

The catch composition for each sector was estimated using the observer data and research logbooks were used to supplement the observer data for the Islander sector. Catch was divided into a number of categories and species groups. Initially catch was divided into four categories; i) overall catch (all landed fish), ii) harvested catch (proportion of the catch retained for commercial sale), iii) bycatch (all species or individuals not retained for commercial sale), and iv) subsistence catch (proportion of bycatch retained by Islander fishers for subsistence use, i.e. not for commercial sale).

Species were grouped into nine categories to allow comparisons between the two sectors (see Appendix 1) including; i) coral trout (*Plectropomus* spp. and *Variola* spp.), ii) barramundi cod (*Cromileptes altivelis*), iii) cods (cods and groupers of the family Serranidae, excluding coral trouts and barramundi cod), iv) emperors (family Lethrinidae), v) snappers (family Lutjanidae), vi) mackerels (family Scombridae), vii) trevallies (family Carangidae), viii) sharks (*Carcharhinus* spp. and *Triaenodon obesus*), and ix) wrasses (family Labridae). Only coral trout were separated further into individual species, as they are the main target species for both sectors.

The catch composition was estimated as the number of individuals in each species group for all data sources and by the total weight of individuals in each species group for the observer survey data. Weight data were only collected for coral trout and barramundi cod from observer surveys. For other species, length data were used to estimate individual fish weights using species specific length-weight relationships obtained from data collected by the Effects of Line Fishing Experiment from the Great Barrier Reef (GBR) (see Mapstone et al., 2004). It was assumed that length-weight relationships for these species were similar between the GBR and ETS.

3.2.6. Analysis of Commercial logbooks and Island Freezer Records

Analysis of data from commercial logbooks and island freezer records focused solely on coral trout. Catch of coral trout is recorded in logbooks and freezer records in a number of product forms including whole, gilled and gutted, and filleted fish. Accordingly, catch was standardised to whole fish weight for both sets of data using specific conversion factors estimated by Begg and Murchie (2004) for each product type.

Effort is recorded at the level of primary vessel days in non-indigenous commercial logbooks, although some records include information on the number of dories used. Effort was estimated in dory days for the non-indigenous sector using the average number of dories per primary vessel for records where the number of dories was not recorded. For Islander fishers, effort is recorded in freezer records at the individual dory level, but no information is available on the number of fishers in each dory (Begg and Murchie 2004). Effort was estimated in Islander days for the Islander sector, which may include more than one fisher per dory.

The standardised catch and effort was used to calculate CPUE. Catch and effort were summed within years and averaged across months to estimate annual and seasonal patterns in catch, effort and CPUE for the two sectors. Analysis of variance (ANOVA) was used to test whether CPUE differed seasonally (month) or annually (year) between the two sectors. CPUE data were truncated to include only years and
months common to both datasets and log-transformed for the analysis to satisfy the assumption of linearity.

Location data for the non-indigenous commercial logbooks were recorded at the scale of individual reef, 6 nautical mile (nm) grid or 30 nm grid. Catch and effort for records that recorded location to an individual reef were assigned to that reef. Catch and effort for records that recorded location to either a 6 or 30 nm grid were distributed proportionately to all the reefs within the grid based on the perimeter of each reef within the grid. The only spatial information available from the island freezer records, however, was the island on which the catch was landed. Therefore, all catch and effort from the Islander sector was distributed to reefs based on trip location data collected from access point surveys of Islander commercial fishers on the islands (S. Busilacchi unpublished data). The proportion of trips by Islander commercial fishers to each reef was used to assign catch and effort to those reefs. A Geographic Information System (GIS) was used to display spatially the average annual catch for each of the sectors.

3.2.7. Analysis of Observer Survey Data
Simple independent sample t-tests were used to compare the following characteristics between the Islander and non-indigenous sectors; i) average fishing time per day, ii) average non-fishing time per day, iii) average ratio of fishing to non-fishing time per day, iv) average fishing time per hang, v) average search time per hang, vi) average time spent anchoring per hang (set time), vii) average distance travelled between consecutive hangs, viii) average depth fished per hang, and ix) CPUE for harvested coral trout, harvested ‘other’ (harvested species other than coral trout species) and bycatch.

CPUE was calculated in kilograms per hour of fishing time for the two harvested categories, while bycatch CPUE was calculated in number per hour of fishing time, as length-weight relationships were unavailable for all bycatch species. This provided standardised units to compare CPUE between the two sectors.

3.3. Results

3.3.1. Catch Composition
A total of 56 species were observed to be caught by Islander and non-indigenous commercial fishers, with a substantial overlap in the species composition between sectors (Appendix 1). Comparison of the overall catch composition by numbers from observer survey data indicated some differences between the Islander and non-indigenous sectors (Fig. 3.1). While coral trout were the predominant species in the catch for both sectors, coral trout comprised a larger proportion of the non-indigenous catch than the Islander catch (Fig. 3.1). Islander fishers caught over twice as many cod but only half the number of snappers as non-indigenous fishers (Fig. 3.1). The relative proportions of the remaining species groups were similar between the two sectors.
Fig. 3.1. Species composition of the total catch in numbers from the Islander (dark shading) and non-indigenous (light shading) sectors of the ETS RLF.

The harvested catch composition from observer survey data also differed between the Islander and non-indigenous sectors (Table 3.1). While non-indigenous fishers harvested approximately 17% more coral trout by number than the Islander fishers, the harvest of coral trout by weight was very similar (Table 3.1). Interestingly, data from the research logbooks, which covered a greater number of Islander fishers and fishing trips, indicated that the non-indigenous sector harvests 27% more coral trout by number than the Islander sector (Table 3.1).

The research logbook data and observer survey data indicated that the Islander sector harvests considerably more cod, emperors and snappers by numbers than the non-indigenous sector (Table 3.1). Comparison of these species groups by weight, however, demonstrated that the proportions of snappers by weight were similar between the two sectors, while the Islander sector harvested less cod by weight than the non-indigenous sector (Table 3.1).

### Table 3.1. Catch composition of harvested catch for the Islander and non-indigenous commercial sectors of the ETS RLF estimated from observer surveys and research logbooks.

<table>
<thead>
<tr>
<th>Species Group</th>
<th>% by numbers</th>
<th>% by weight</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Islander</td>
<td>Non-indigenous</td>
</tr>
<tr>
<td>Coral trout</td>
<td>65.41%</td>
<td>70.67%</td>
</tr>
<tr>
<td>Snappers</td>
<td>15.76%</td>
<td>12.37%</td>
</tr>
<tr>
<td>Emperors</td>
<td>7.61%</td>
<td>6.71%</td>
</tr>
<tr>
<td>Cod</td>
<td>3.59%</td>
<td>4.24%</td>
</tr>
<tr>
<td>Trevally</td>
<td>1.96%</td>
<td>1.06%</td>
</tr>
<tr>
<td>Mackerel</td>
<td>4.07%</td>
<td>2.47%</td>
</tr>
<tr>
<td>Barramundi cod</td>
<td>0.97%</td>
<td>1.77%</td>
</tr>
<tr>
<td>Wrasse</td>
<td>0.58%</td>
<td>0.71%</td>
</tr>
<tr>
<td>Shark</td>
<td>0.04%</td>
<td>0.00%</td>
</tr>
</tbody>
</table>
The species composition of harvested coral trout differed between the two sectors (Fig. 3.2). While the majority (approximately 80%) of coral trout catch by numbers was comprised of *P. leopardus* for both sectors, the proportion of other coral trout species differed between sectors (Fig. 3.2). The non-indigenous sector harvest proportionally more *P. maculatus* and *P. areolatus* than Islander fishers. However, Islander fishers harvested proportionally more *P. laevis* than non-indigenous fishers, and were the only sector to record catches of *Variola* spp. (Fig. 3.2).

**Fig. 3.2.** Species composition of the harvested catch of coral trout in numbers from the Islander (dark shading) and non-indigenous (light shading) sectors of the ETS RLF.

Bycatch was a significant component of the catch from both sectors. The number of species that were never harvested was greater for non-indigenous fishers (59.6%) than for Islander fishers (38.9%) (Appendix 1). The proportion of bycatch in the catch by number of individual fish, however, was similar between Islander fishers (57.8%) and non-indigenous fishers (54.9%) (Fig. 3.3). Individuals of harvested species that were smaller than the legal minimum size also formed part of the bycatch for both sectors (Appendix 1).
The subsistence catch made up 7.6% of the overall Islander catch by numbers and 13.2% of the Islander bycatch by numbers (Fig. 3.3). Differences also existed in the bycatch species composition between the two sectors (Fig. 3.4). The majority of Islander fishers’ bycatch was comprised of cod while the non-indigenous bycatch was comprised mainly of snappers, cods, emperors and less than legal size coral trout (Fig. 3.4). More than half of the snapper bycatch, all of the trevally and a small proportion of the emperor and coral trout bycatch from the Islander fishers were retained as subsistence catch (Fig. 3.4).
3.3.2. Commercial catch and effort data

The trend in annual catch and CPUE for coral trout from the commercial Islander and non-indigenous sectors varied over the years (Fig. 3.5). Data for 1988 from the non-indigenous sector are not presented here, as the data were incomplete for that year. Annual catch for the non-indigenous sector has increased since 1989, reaching a number of peaks in 1994, 1998, 2001 and 2003 (Fig. 3.5a). The peak of more than 144 t in 2001 was the highest recorded for this sector. CPUE for the non-indigenous sector has been relatively stable over most years except for a strong peak in 2000 and 2001. The annual catch for the Islander sector appears to have increased substantially since 1989, reaching a peak of nearly 30 t in 2004 (Fig. 3.5b). It must be noted, however, that this apparent increasing trend in catch is most likely due to the unavailability of records, particularly prior to 2003. CPUE for the Islander sector was highly variable, but the highest CPUE occurred in the same years as for the non-indigenous sector (Fig. 3.5).

**Fig. 3.5.** Annual catch (bars) and CPUE (lines) for coral trout estimated from commercial logbook data and island freezer records for the (a) non-indigenous and (b) Islander commercial sectors of the eastern Torres Strait reef line fishery. Note different y-axis scales.
Seasonal patterns in catch and effort revealed clear differences in the harvest patterns for coral trout between Islander and non-indigenous commercial sectors (Figs. 3.6a and b). Catch and effort were generally highest from October to March for Islander fishers, while non-indigenous catch and effort were generally highest from March to October (Figs. 3.6a and b). The period of lowest catch and effort for Islander fishers was during April to September, while non-indigenous catch and effort was lowest from December to February (Figs. 3.6a and b). Differences in the seasonal patterns in CPUE between the two sectors were less clear than for catch and effort, but CPUE was highest in August for Islander fishers and in March for non-indigenous fishers (Fig. 3.6c). ANOVA revealed a significant interaction between the effects of year, month and sector on CPUE ($F = 3.672$, $p < 0.001$) indicating that catch rates vary depending on the year, month or sector. In general, however, the non-indigenous sector has consistently harvested substantially more coral trout per year than the Islander sector (Fig. 3.5). The average CPUE has also been consistently higher for the non-indigenous sector than the Islander sector in all years (Fig. 3.5) and months (Fig. 3.6c).
Fig. 3.6. (a) Average monthly catch, (b) average monthly effort and (c) average monthly CPUE estimated from commercial logbook data and island freezer records for the non-indigenous (dashed line) and Islander (solid line) commercial sectors of the ETS RLF. Note different y-axis scales.

The spatial distribution of the average annual commercial catch of coral trout for the two sectors revealed that the non-indigenous catch (Fig. 3.7) is distributed over more reefs and a much larger area than the Islander catch (Fig. 3.8). While the Islander catch is concentrated on reefs near the islands (Fig. 3.8), the non-indigenous catch is concentrated on a group of large reefs associated with the Cumberland Passage (Fig. 3.7).
**Fig. 3.7.** Spatial distribution of the average annual coral trout catch in the ETS estimated from commercial logbook data for the non-indigenous commercial sector of the RLF.
3.3.3. Observer Survey Data

CPUE from observer survey data was consistently higher for non-indigenous fishers than Islander fishers for all catch categories (Fig. 3.9). CPUE differed significantly between the two commercial fishing sectors for harvested coral trout only ($t = 2.67, p < 0.01$), while CPUE was not significantly different between the two sectors for harvested ‘other’ species ($t = 1.08, p = 0.28$) or bycatch ($t = 0.96, p = 0.34$).
Islander fishers spent on average more time fishing per day than non-indigenous fishers (Fig. 3.10a), although this difference was not significant ($t = 1.26$, $p = 0.22$). The average non-fishing time per day did vary significantly between the two sectors ($t = 2.12$, $p < 0.05$), with non-indigenous fishers spending on average more time not fishing per day than Islander fishers (Fig. 3.10b). The ratio of fishing to non-fishing time per day also varied significantly between the two sectors ($t = 2.31$, $p < 0.05$), with a higher ratio of fishing to non-fishing time per day for Islander fishers (Fig. 3.10c).
Fig. 3.10. (a) Average fishing time per day, (b) average non-fishing time per day, and (c) average ratio of fishing to non-fishing time per day from observer survey data for the Islander and non-indigenous sectors of the ETS RLF. Error bars are standard error.

Fishing time per hang differed significantly between the two sectors ($t = 4.24, p < 0.001$), with Islander fishers spending on average more time fishing per hang than non-indigenous fishers (Fig. 3.11a). Non-indigenous fishers spent on average more search time and set time per hang than Islander fishers (Figs. 3.11b and c), but neither the search time ($t = 0.41, p = 0.68$) nor set time ($t = 0.80, p = 0.43$) were significantly different between sectors.
The average distance travelled between consecutive hangs did not differ significantly between sectors ($t = 0.25$, $p = 0.80$, Fig. 3.12a). Average hang depth differed significantly between the two sectors ($t = 2.21$, $p < 0.05$), with Islander fishers fishing deeper water on average than non-indigenous fishers (Fig. 3.12b). The average number of non-indigenous fishers per dory was 1 with no error, while the average number of Islander fishers per dory was 1.36 ($\pm 0.15$ s.e.).
Fig. 3.12. (a) Average distance travelled between consecutive hangs, and (b) average depth per hang from observer survey data for the Islander and non-indigenous sectors of the eastern Torres Strait reef line fishery. Error bars are standard error.

3.4. Discussion

The harvest patterns and effort dynamics differed significantly between the Islander and non-indigenous sectors of the commercial RLF in the ETS. Differences between sectors were observed in species composition and spatial and temporal patterns in catch, effort and CPUE. These results highlight the inherent variation in the catch and effort dynamics between the two commercial sectors of the ETS RLF, and provide crucial information which should be incorporated into future assessments of the fishery and considered in developing appropriate management strategies.

A number of factors may have contributed to the observed differences in catch and catch rates of coral trout between the Islander and non-indigenous sectors. Non-indigenous fishers harvested substantially more coral trout each year than Islander fishers. The proportion of coral trout in the catch and CPUE for coral trout were also consistently higher for non-indigenous fishers than for Islander fishers, suggesting potential differences in fisher behaviour between sectors, or a greater efficiency of non-indigenous fishers to target coral trout. Different motivations of Islander and non-indigenous fishers are likely to be a primary reason for differences in catch and CPUE. Maximising economic return is the primary motivation for the non-indigenous commercial sector, while lifestyle and meeting more immediate needs, rather than long-term financial planning, is the primary motivation for the Islander commercial sector (Scott and Mulrennan 2003, Arthur 2005). Differences in fishing skill level between Islander and non-indigenous fishers are also likely to contribute to
differences in catch and CPUE between the two sectors. Non-indigenous fishers have been line fishing for reef fish in Torres Strait from powered dories since the 1950's (Haysom 2001). In contrast, Islander fishers are generally less experienced in using these fishing techniques and the skills have been transferred to Islander commercial fishers only recently (Altman et al. 1994). The higher catch and CPUE for non-indigenous fishers may also be due to the large mobile primary vessels used by non-indigenous fishers which allow them to move to other areas of potentially higher catch rates when low catch rates are experienced. Islander fishers are more restricted in the area in which they can fish (Altman et al. 1994), given the limitations of their smaller vessels (Scott and Mulrennan 2003), and therefore do not have access to as many reefs as non-indigenous fishers. Islander fishers are therefore likely to return more often to reefs that have produced lower catch rates.

The spatial and temporal patterns in catch and effort varied substantially between sectors. Non-indigenous fishers operated over a much wider area than Islander fishers, including all reefs fished by Islander fishers. Mapstone et al. (2003) reported a similar distribution of non-indigenous catch and effort, but at a more coarse spatial scale. A large proportion of catch from non-indigenous fishers, however, arose from reefs south of the Cumberland Passage that are not frequently fished by Islander fishers, suggesting that non-indigenous fishers may be harvesting a different component of the resource than Islander fishers. Furthermore, the months of peak catch and effort did not overlap between sectors, indicating the existence of specific fishing seasons for each sector. Islander fishers are more active between October and March when there are breaks in the south-east trade winds that allow them to travel from the islands across open areas to more distant reefs (Begg and Murchie 2004). Non-indigenous fishers are not as restricted by weather, as they are able to seek refuge behind large reefs during strong winds where they are able to fish with relative protection. Most non-indigenous fishers leave the ETS in November or December each year, as the barges that service the ETS cease operation over the summer months. Non-indigenous fishers usually return to the ETS in February or March when the barges commence operation (Begg and Murchie 2004). This spatial and temporal partitioning of catch and effort between the Islander and non-indigenous commercial sectors may reduce the potential for conflict between sectors over competition for the resource (Begg and Murchie 2004).

The fine scale catch and effort data collected through observer surveys allowed direct comparison of the effort dynamics and efficacy of the two commercial sectors in the ETS RLF. CPUE was higher for non-indigenous fishers than Islander fishers for all catch categories, even though the Islander sector had, on average, more fishers per dory than the non-indigenous sector. This finding is likely to be reflective of the differences in fisher behaviour between the two sectors. Non-indigenous fishers spent less time fishing at a hang and more time searching and setting each hang than Islander fishers. It seems likely that more time taken in carefully selecting and positioning each hang resulted in better catch rates for the non-indigenous fishers. The shorter time spent fishing at each hang by non-indigenous sector suggests that they do not wait for catch rates to increase on hangs with poor catch rates and leave a hang once catch rates decline. The shallower water fished by non-indigenous fishers, compared with Islander fishers, may also have resulted in higher CPUE, as non-indigenous fishers are able to retrieve and return each line more quickly. Furthermore, the greater mobility of the non-indigenous sector, due to the large primary vessels, enables non-indigenous fishers to relocate to different reefs in the same day if low catch rates are experienced. This finding, coupled with the mid-day break from fishing taken by non-indigenous fishers, meant that non-indigenous fishers spent less time on average per day fishing than Islander fishers.
Differences in the fine scale effort dynamics between the two sectors suggests that effort levels differ between them. It is therefore likely that different levels of effort are required by the different sectors to harvest the same catch quantity, as CPUE for coral trout estimated from observer survey data differed between the two sectors. The comparison of CPUE between the commercial logbook data for the non-indigenous sector and freezer records data for the Islander sector is unlikely to be valid, given the different units of effort associated with the different datasets. This has significant implications for the management of the ETS RLF, as management is currently undergoing a shift towards a quota based management system using individual transferable units of catch or effort (AAP 2006). As the allocation of these units will be based on presumed levels of sustainable catch (AFMA 2005a), accurate knowledge of the catch rates for each sector will be essential in the allocation of effort units for each sector.

The species caught by both sectors were very similar (Appendix 1), indicating that both sectors are sharing the same resource in the ETS. The relative proportion of each species in the catch, however, differed between the two sectors, which may be explained by differences in targeting behaviour, species distribution, or both. Islander fishers harvested more cods, snappers and emperors than non-indigenous fishers. However, the average size of the cods and snappers were substantially smaller for Islander fishers than for non-indigenous fishers. This most likely reflects a difference in the species composition of these harvested families, with Islander fishers harvesting more of the smaller species such as *Epinephelus ongus* and *Lutjanus carponotatus* and non-indigenous fishers harvesting more of the larger species such as *E. fuscoguttatus* and *L. sebae* (Appendix 1). The lower proportion of *P. maculatus* and *P. areolatus* in the Islander harvest, however, is most likely due to the restriction of observer surveys to Murray Island fishers and the majority of the research logbook data coming from the Murray Island freezer. While *P. leopardus* is broadly distributed throughout the ETS, *P. maculatus* and *P. areolatus* are most abundant on western reefs, particularly in the north between Darnley and Yorke Islands, but are rare on reefs closer to Murray Island (see Chapter 4). Preliminary analyses indicate that *P. maculatus* and *P. areolatus* are much more common in catches from Islander commercial fishers on Darnley Island (Williams unpublished data). However, as the majority of the Islander sectors catch comes from Murray Island (Begg and Murchie 2004), the composition of the catch estimated from the research logbook data is likely to be somewhat reflective of the Islander sectors catch composition.

Bycatch contributed a significant proportion (55-58% by numbers and 38-59% by number of species) to the overall catch for both sectors of the ETS RLF. This finding is contrary to earlier views that there are relatively low levels of bycatch in the fishery (Mapstone et al. 2003, AFMA 2005a, AFMA 2005b). Earlier perceptions were based on the assumption that the fishery targeted multiple species and that most species caught were retained (AFMA 2005b). Clearly, our results indicate that bycatch is a significant component in the fishery. Subsistence catch by Islander fishers also made up a small, albeit important, proportion of the bycatch. Current estimates of catch for the Islander sector of the ETS RLF do not include reef fish harvested for subsistence (Begg and Murchie 2004), and therefore total catch for the Islander sector will be underestimated if subsistence catch is not considered. It will be important to determine the survival rates of released bycatch species in the fishery, in addition to the estimates of the abundance of bycatch species in the catch provided in this study. This will allow the relative impact of the fishery on each species to be assessed so that potentially vulnerable species can be identified. Appropriate management measures may need to be implemented for species identified as being at high risk from fishing.
Barramundi cod (*Cromileptes altivelis*) and Maori wrasse (*Cheilinus undulatus*) are two species currently caught in the ETS RLF that are listed on the IUCN Red List of threatened species. These two species are now protected on the GBR having recently been legislated as no-take. During the period of this study, Maori wrasse was harvested legally in Torres Strait, but in September 2006, they became a no-take species. Maori wrasse has also been listed on the CITES trade list due to concerns of declining wild populations from overexploitation, particularly from the international live fish food trade (IUCN Groupers & Wrasses Specialist Group 2006). Little is known about the biology of these species in Torres Strait (but see Chapter 5 for information on barramundi cod), and given the new legislative requirements to assess and manage the risk to each species (Fletcher 2003), more detailed information on the biology and catch of these species in the ETS RLF is urgently needed.

The multi-species nature of the ETS RLF has important implications for quota based management given the issues and difficulties associated with managing multi-species fisheries (Squires et al. 1998, Sanchirico et al. 2006). Problems with assessment and management arise as the complexity of the fishery increases (Cochrane 1999), which is the case when multiple sectors are operating within a multi-species fishery. These problems are likely to be further exacerbated when the different sectors have different harvest patterns and effort dynamics, as we have demonstrated with the Islander and non-indigenous commercial sectors of the ETS RLF. It will be important, therefore, to consider these sector-specific factors for sustainable management of the fishery.

This study demonstrated substantial differences in harvest patterns and effort dynamics between the Islander and non-indigenous commercial sectors of the ETS RLF. Observer surveys provided valuable information on species composition of the catch and bycatch, and more reliable estimates of catch, effort and CPUE for each sector, which has not been available previously for this fishery. This information will assist with the development of a formal stock assessment for the fishery and will be essential for management of the ETS RLF as it heads towards a quota based management system. Furthermore, this information will be useful for resolving issues relating to allocation of reef fish resources in Torres Strait and for achieving federal legislative requirements for sustainable use and management of Australian fisheries (Fletcher 2003, Lenanton 2003).
4. Population biology of coral trout species in the eastern Torres Strait: Implications for fishery management

4.1. Introduction

Coral reefs support many important commercial, recreational and subsistence fisheries, particularly around the large population centres of the islands of the Caribbean and south-east Asia. In these areas, rapidly expanding populations coupled with inadequate management have placed increasing pressure on reef fish populations, and many fisheries are now considered to be over-exploited (Russ 1991, Sale 2002, Sadovy 2005). Managing coral reef fisheries is complex, however, as most reef fisheries are multi-species and multi-gear, capturing a large number of species with a range of gears including hook and line, spears, traps, nets, poisons and explosives (Dalzell 1996).

The complexity of managing coral reef fisheries is exacerbated by the diverse range of life histories of coral reef fish (Sale 1991), and their inter and intra-specific interactions. For example, relatively small species such as caesionids that have high natural mortality rates and a maximum longevity of only 2-3 years (Carpenter 2001) may be harvested in the same fishery as larger species such as epinepheline serranids and lutjanids that have relatively low natural mortality rates and potential longevities in excess of 40 years (Wilson and Nieland 2001, Pears et al. 2006). Sexual strategies may also vary among species harvested in the same fishery. For example, serranids are typically protogynous hermaphrodites, changing sex from female to male, while lutjanids are gonochoristic, remaining the same sex throughout life (Sadovy 1996). Clearly, population dynamics are likely to differ among target species, as is their response to varying fishing patterns (Dugan and Davis 1993). In order to improve management of coral reef fisheries it is important to understand the dynamics of reef fish populations, the processes that control them, and their likely responses to different levels of fishing pressure. Unfortunately, research on the population dynamics of reef fish in the past generally has been focussed on non-harvested rather than harvested species, and consequently our knowledge of the dynamics of exploited reef fish populations is relatively limited (Roberts 1996).

Coral trout (Plectropomus spp.) are the target species for both commercial sectors of the ETS RLF. There are four species of coral trout known to occur in Torres Strait; common coral trout (P. leopardus), bar-cheeked trout (P. maculatus), passionfruit trout (P. areolatus) and footballer or blue-spot trout (P. laevis) (Mapstone et al. 2003). Currently, there is no biological information available for any of these species in Torres Strait. Information on the biology of coral trout is available from other locations, but mainly for P. leopardus (Ferreira 1995, Ferreira and Russ 1995, Adams et al. 2000), P. maculatus (Ferreira and Russ 1992, Ferreira 1993, Adams and Williams 2001) and P. laevis (Davies et al. 2006) and predominantly from research on the Great Barrier Reef (GBR). There is very limited knowledge on the biology of P. areolatus from any location. The presence of substantial regional variation in reef fish population biology (e.g. Adams et al. 2000, Williams et al. 2003) means that it is unclear whether information on coral trout biology from other locations is transferable to the Torres Strait. Developing an understanding of the population biology of coral trout species in Torres Strait, therefore, has been identified as a priority research area for sustainable management of the ETS RLF (Mapstone et al. 2003).

Currently, all species of coral trout are managed as a single species in the ETS RLF, with a single minimum size limit of 38 cm total length (TL). However, available
information from other locations has demonstrated substantial variation in biology among coral trout species (e.g. Adams 2003, Davies et al. 2006), suggesting that different species may respond differently to fishing. The aim of our study was to determine whether it is appropriate to manage coral trout in the ETS RLF as a single species group, based on the biology of each species, or whether different management arrangements are required for each species. To achieve this we examined and compared the distribution, age, growth, mortality, length and age at sex change, sex ratios and spawning season for each coral trout species in the ETS.

4.2. Materials and Methods

4.2.1. Sampling Methods

Information on the catch composition of coral trout species was obtained during observer surveys on board Islander and non-indigenous commercial vessels operating in the ETS (see Chapter 2). Biological samples of P. leopardus, P. maculatus, P. areolatus and P. laevis were collected during the observer surveys on board non-indigenous commercial vessels. During the surveys, biological samples of P. leopardus, P. maculatus, P. areolatus and P. laevis were collected and fork length (FL) and whole wet weight (W) were measured for all coral trout species caught. Sagittal otoliths (hereafter otoliths) and gonads were removed from all coral trout up to a maximum of 100 individuals of each species per trip.

Additional biological samples of P. leopardus, P. maculatus and P. areolatus were collected in the form of frozen fish frames (head, skeleton and viscera) from non-indigenous commercial fishers from March 2004 to November 2005. Samples were not available from all months in each year due to the opportunistic nature of collecting samples from commercial fishers. The FL was measured and otoliths and gonads removed from each individual. Length and sex information was not available from all samples as the gonads or skeleton were occasionally missing.

4.2.2. Sample Processing

All otoliths were cleaned and stored dry in small plastic vials. Otoliths were embedded in clear casting polyester resin and cut transversely at 300-400 µm through the core using twin diamond blades on a low speed diamond-blade saw. Sections were embedded on slides using the resin and covered with cover slips. Opaque increments viewed under reflected light were counted from the nucleus to the edge of the otolith along the ventral margin of the sulcus acousticus using a stereo dissecting microscope at 40× magnification. The opaque increment closest to the margin of the otolith was only included in the count when the reader interpreted the increment as completely formed. All sectioned otoliths were read twice by a single experienced reader. If the counts from the two readings were the same, the age was accepted. If the counts differed, a third count was made by the same reader. If the third count matched either the first or second count, that count was accepted. There were 20 P. leopardus, 12 P. areolatus and 8 P. maculatus for which there was no agreement among three counts. For these otoliths, the median of three age estimates was assigned as the final age.

Because samples were collected across different months of the year, it was necessary to adjust age estimates to account for the time of increment formation and time of capture. The following algorithms were used to adjust age estimates:

\[ \text{age}_m = m_c + 12 \quad \text{if } N = 0, \quad (4.1) \]

\[ \text{age}_m = (N \times 12) + m_b + m_c \quad \text{if } N > 0 \quad (4.2) \]
where $age_m$ is the age in months, $N$ is the number of complete opaque increments, $m_b$ is the number of months from assigned birth date to time of opaque increment completion, and $m_c$ is the number of months from time of opaque increment completion to time of capture. The date of opaque increment completion was assigned at October 1 for all species and was based on the period of annulus formation described for $P. \text{leopardus}$ from the GBR (Ferreira and Russ 1994). The assigned birth date was December 1 for $P. \text{leopardus}$ and $P. \text{maculatus}$ and September 1 for $P. \text{areolatus}$ based on knowledge of the spawning season from our study. Adjusted age estimates were rounded to the nearest year for estimation of age frequency distributions, total mortality rates and age at sex change.

Gonads were frozen immediately after removal and transported back to the laboratory where they were thawed and preserved in a solution of 10% buffered formalin. After fixation, each pair of ovaries or testes was dried of excess fixative, and gonad weight was measured to the nearest 0.01 g. Where only one gonad lobe was available, due to damage during processing, gonad weight was estimated by multiplying the mass of the single complete lobe by two. It was assumed that this provided a reasonably accurate measure of gonad weight, as the two gonad lobes are generally equal in size for coral trout (Adams 2003). Gonad weight was estimated using this method for 90 $P. \text{leopardus}$, 23 $P. \text{maculatus}$ and 27 $P. \text{areolatus}$.

Histological sections were taken from all gonads following the procedures outlined by Adams (2003). Ovaries were staged based on the most advanced non-atretic cell type present (West 1990). Additional features used in histological staging included the presence of brown bodies, atretic oocytes, vascularisation, and the relative thickness of the gonad wall, all of which may indicate prior spawning (Sadovy and Shapiro 1987). Ovaries and testes were classified into developmental stages adapted from Ferreira (1995) and Adams (2003). Females were classified into four stages: Immature, Resting, Ripe and Hydrated. Males were classified into three stages: Spent, Resting and Ripe. Transitional (fish in the process of changing sex from female to male) and bisexual (fish able to function as female and male simultaneously) individuals were also identified using the criteria described by Sadovy and Shapiro (1987).

### 4.2.3. Distribution

The distribution of each coral trout species in the ETS was estimated from the catch of both Islander and non-indigenous fishers. A Geographic Information System (GIS) was used to plot the relative proportion of each coral trout species caught from each reef sampled in the study area of the ETS (see Fig. 2.1), which provided an approximation of the distribution for each species.

### 4.2.4. Length-weight relationship

The relationship between $FL$ and $W$ was described for each coral trout species by a power function of the form,

$$ W = a \times FL^b $$  \hspace{1cm} (4.3)

where $a$ is the coefficient of the power function and $b$ is the exponent. This relationship was compared among species using ANCOVA with $FL$ the covariate of $W$. Length and weight data were log-transformed for the analysis to satisfy the assumption of linearity. Sources of inferred differences in the slope ($b$) or intercept ($a$) of the log-transformed relationship were identified by Bonferroni adjusted multiple comparisons (Milliken and Johnson 2001).
4.2.5. Length and Age frequency distributions

Length and age frequency distributions were constructed for each coral trout species. Multi-dimensional contingency tables were then used to compare length and age frequency distributions among species. Age classes 2 years and younger and age classes 13 years and older were pooled into 2- and 13+ age classes due to low frequencies in the tails of the age distributions and to ensure that the same range of age classes were compared among species. Similarly, length classes 310 mm FL and smaller and length classes 650 mm FL and larger were pooled into 310- mm and 650+ mm length classes.

4.2.6. Growth

The parameter estimates from the Schnute (1981) growth function suggest the von Bertalanffy growth function (VBGF) provides the best fit to length-at-age data for P. leopardus (Ferreira and Russ 1994), and was the model used here for consistency among species. The VBGF was fitted by nonlinear least-squares regression of FL on age for each species. The form of the VBGF used to model length-at-age data was,

\[ L_t = L_\infty \left(1 - e^{-K(t-t_0)}\right) \]  

where \( L_t \) is the length-at-age \( t \), \( L_\infty \) is the mean asymptotic fork length, \( K \) is the growth coefficient or rate at which \( L_\infty \) is approached, \( t \) is the age of the fish and \( t_0 \) is the age at which the fish have a theoretical length of zero.

VBGF’s were compared among species using likelihood ratio tests (Kimura 1980), which are considered the most reliable procedure for such comparisons (Cerrato 1990). A common range of age classes was used in each analysis to assure validity of the comparisons (Haddon 2001).

Differences in growth among species were further examined by comparing the approximate 95% confidence ellipses for the VBGF parameter estimates of \( L_\infty \) and \( K \) among species (Draper and Smith 1966, Kimura 1980). Using this technique to compare growth parameters requires the VBGF parameter \( t_0 \) to be constrained to the same value for all growth curves being compared. A common value for \( t_0 \) was obtained for each species by fitting an unconstrained VBGF to the length-at-age data from all species. The constraining of \( t_0 \) usually alters estimates of \( L_\infty \) and \( K \) due to the correlation among VBGF parameters (e.g. Pilling et al. 2002). Consequently, the results from this technique for growth comparison are not always consistent with results from the likelihood ratio tests. Furthermore, estimates of the differences among growth curves will be more conservative (i.e., less likely to be different) when \( t_0 \) is constrained to the same value for all curves due to the correlation among VBGF parameters.

4.2.7. Mortality

Age-based catch curves (Ricker 1975) were used to estimate the instantaneous rate of total mortality (Z) for each species. The number of fish in each age class was regressed against the corresponding age, and the descending slope provided an estimate of Z. Regressions were fitted from the first age class that was fully selected by the sampling gear through to the oldest age class that was preceded by no more than two consecutive zero frequencies. Mortality rates were compared among species using ANCOVA, with age as the covariate. Sources of inferred differences in the slope (b) of the regression were identified by Bonferroni adjusted multiple comparisons (Milliken and Johnson 2001).
4.2.8. Spawning seasonality

The spawning season for each coral trout species was estimated from samples collected during observer surveys on commercial non-indigenous vessels and from monthly fish frames collected from the non-indigenous commercial sector. A gonadosomatic index (GSI: gonad weight / W x 100) was calculated for each sample, which provided a relative measure of reproductive stage. Measurements of W were not available directly from fish frames. Consequently, W was estimated for these samples using the species-specific length-weight relationship. It was assumed that estimates of W would suffice for estimates of GSI because only the temporal patterns in GSI, rather than the absolute values of GSI, were required to determine the spawning season. The proportion of samples in each mature female and male reproductive stage in each month were also plotted for each species to examine the ovarian and testicular development patterns throughout the year, and the degree of spawning activity occurring in each month for each species.

4.2.9. Length and age at sex change

A logistic function was fitted to the proportion of males in each length and age class to estimate the length and age at which each species of coral trout changes sex. The length and age at sex change was estimated for each species using the logistic equation,

\[ P_s = \left(1 + e^{-\ln(19)(s-s_{50})/(s_{95}-s_{50})}\right)^{-1} \]  

(4.5)

where \( P_s \) is the proportion of males in age or 10 mm length class \( s \), and \( s_{50} \) and \( s_{95} \) are the age or length at which 50% and 95% of the population are males for each species, respectively. The presence or absence of transitional and bisexual individuals in each length and age class also provided an estimate of the length and age range over which sex change occurs. Likelihood ratio tests were used to test for differences in the length and age at sex change among species using a common age or length range for each comparison to assure validity of the comparisons (Haddon 2001).

The sex assigned from histology for samples collected during observer surveys was used to calculate the sex ratios for each species, which were compared between species using a chi-square contingency test and compared with an expected ratio of 1:1 by a \( \chi^2 \) goodness of fit test.

4.3. Results

The number of biological samples of \( P. leopardus \) (N = 689), \( P. maculatus \) (N = 336) and \( P. areolatus \) (N = 452) collected was sufficient to estimate biological parameters for each of these species. However, only 25 \( P. laevis \) were sampled which was insufficient to estimate any biological parameter. Accordingly, estimates of biological parameters are presented for only three species of coral trout.

4.3.1. Distribution

The distribution of coral trout species, estimated from the proportion of each species recorded in the catch from each reef, varied substantially among species (Fig. 4.1). \( P. leopardus \) was caught from every reef that was sampled and, therefore, appears to have a broad distribution throughout the ETS. In contrast, \( P. maculatus \) and \( P. areolatus \) were not caught on all reefs sampled. They tended to be more common on the western reefs, particularly in the north, where on some reefs they were the most abundant coral trout species caught. The few \( P. laevis \) samples collected were
caught from reefs near the outer barrier reefs in the east, suggesting a more restricted outer-reef distribution.

Fig. 4.1. Estimated distribution of *P. leopardus*, *P. maculatus*, *P. areolatus* and *P. laevis* in the eastern Torres Strait. Pie charts indicate the proportion of each species recorded in commercial catches from each reef fished.

4.3.2. Length-weight relationship

The relationship between *FL* and *W* was approximately isometric \((b \approx 3)\) for each coral trout species (Fig. 4.2, Table 4.1) indicating that each species becomes more heavy-bodied as they grow. The slope of the *FL*-*W* relationship \((b)\) did not differ significantly among species \((F = 2.19, p > 0.1)\). However, the intercept (or elevation) of the relationship differed significantly among species \((F = 11.45, p < 0.001)\). The intercept for *P. areolatus* was significantly greater than the intercept for *P. leopardus* and *P. maculatus* (Table 4.1), indicating that *P. areolatus* were generally heavier for a given length than either *P. leopardus* or *P. maculatus*. 
Fig. 4.2. Length at weight data and fitted power curves for (a) *P. leopardus*, (b) *P. maculatus* and (c) *P. areolatus* from the eastern Torres Strait.

<table>
<thead>
<tr>
<th>Species</th>
<th>(a\times10^{-6})</th>
<th>(b)</th>
<th>(L_\infty)(mm)</th>
<th>(K)</th>
<th>(t_0) (yrs)</th>
<th>(Z) (yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>P. leopardus</em></td>
<td>2.36</td>
<td>3.30</td>
<td>746</td>
<td>0.07</td>
<td>-7.20</td>
<td>0.44</td>
</tr>
<tr>
<td><em>P. maculatus</em></td>
<td>3.85</td>
<td>3.22</td>
<td>687</td>
<td>0.09</td>
<td>-6.33</td>
<td>0.61</td>
</tr>
<tr>
<td><em>P. areolatus</em></td>
<td>2.91</td>
<td>3.27</td>
<td>764</td>
<td>0.09</td>
<td>-5.87</td>
<td>0.40</td>
</tr>
</tbody>
</table>

Table 4.1. Parameter estimates for the length-weight relationship, VBGF and rates of total mortality for three species of coral trout from the ETS. \(a\) and \(b\) are parameters of the allometric relationship (\(W = a \times FL^b\)) between fork length (\(FL\)) and weight (\(W\)). \(L_\infty\) is the mean asymptotic fork length, \(K\) is the von Bertalanffy growth coefficient, and \(t_0\) is the theoretical age at length zero. \(Z\) is the instantaneous rate of total mortality.
4.3.3. Length and age frequency distributions

The modal age in the catch was four years for *P. leopardus* and five years for *P. maculatus* and *P. areolatus*, although there was a second equivalent mode of seven years for *P. areolatus* (Fig. 4.3). The maximum age was similar among species with the oldest fish being a 17yo *P. leopardus* (Fig. 4.3). Multi-dimensional contingency tables indicated that the age distributions varied significantly among all coral trout species ($\chi^2 = 88.94, p < 0.001$). Pairwise comparisons revealed that age distributions varied significantly between each pair of species (*P. leopardus* vs. *P. maculatus*: $\chi^2 = 26.63, p = 0.009$; *P. leopardus* vs. *P. areolatus*: $\chi^2 = 38.11, p < 0.001$; *P. maculatus* vs. *P. areolatus*: $\chi^2 = 69.93, p < 0.001$). The most noticeable difference among species was the greater relative abundance of older (>7 years) *P. areolatus* compared with the other two species (Fig. 4.3).

Fig. 4.3. Length and age frequency distributions for (a) *P. leopardus*, (b) *P. maculatus* and (c) *P. areolatus* from the ETS.
The modal length in the catch differed among coral trout species (Fig. 4.3) and was largest for *P. areolatus* (490 mm FL), smallest for *P. maculatus* (390 mm FL) and intermediate for *P. leopardus* (430 mm FL). The maximum length, however, was very similar among species (Fig. 4.3). Multi-dimensional contingency tables indicated that the length distributions varied significantly among all coral trout species ($\chi^2 = 154.14$, $p < 0.001$). Pairwise comparisons revealed that the length distribution for *P. areolatus* differed significantly from both *P. leopardus* ($\chi^2 = 123.22$, $p < 0.001$) and *P. maculatus* ($\chi^2 = 97.80$, $p < 0.001$), but did not differ significantly between *P. leopardus* and *P. maculatus* ($\chi^2 = 22.85$, $p = 0.15$). The most noticeable difference among species was the greater relative abundance of larger (>490 mm FL) *P. areolatus* compared with the other two species (Fig. 4.3).

### 4.3.4. Growth

There was substantial variation in the length-at-age of all coral trout species, particularly *P. leopardus* and *P. maculatus* (Fig. 4.4). The fitted growth curves were relatively ‘flat’, as few fish smaller than 300 mm FL were sampled. Therefore, growth curves were not generally of asymptotic form and estimates of VBGF parameters were most likely biased. Nevertheless, comparisons of growth curves among species are still valid assuming the selectivity bias is similar for all species and that comparisons are made across the same age range (Haddon 2001).

![Fig. 4.4. Length-at-age data and fitted VBGF curves for (a) *P. leopardus*, (b) *P. maculatus* and (c) *P. areolatus* from the ETS. (d) Approximate 95% confidence ellipses of the VBGF parameters $L_\infty$ and $K$ for each species.](image)

Likelihood ratio tests indicated that patterns of growth differed significantly among coral trout species ($\chi^2 = 50.76$, $p < 0.001$). The growth pattern for *P. areolatus* was significantly different to *P. leopardus* ($\chi^2 = 61.52$, $p < 0.001$) and *P. maculatus* ($\chi^2 = 9.64$, $p = 0.02$). The growth pattern did not differ significantly between *P. leopardus* and *P. maculatus* ($\chi^2 = 7.76$, $p = 0.05$). The most noticeable difference in growth was that *P. areolatus* reached a larger average maximum length ($L_\infty$) than both *P.*
leopardus and P. maculatus (Table 4.1). The poor fit of the growth curves is reflected in the size of the 95% confidence ellipses for the VBGF parameters $L_\infty$ and $K$, particularly for P. maculatus (Fig. 4.4). There was no overlap in estimates of the 95% confidence ellipses between P. leopardus and P. areolatus (Fig. 4.4), which is consistent with the results of the likelihood ratio tests. However, there was overlap between the confidence ellipse for P. maculatus and the other two species, which was not consistent with the results of the likelihood ratio tests.

### 4.3.5. Mortality

Estimates of total mortality ($Z$) differed among coral trout species (Fig. 4.5) and were highest for P. maculatus, lowest for P. areolatus and intermediate for P. leopardus (Table 4.1). However, ANCOVA indicated that $Z$ did not differ significantly among coral trout species ($F = 3.32, p = 0.05$).

![Catch curves](image)

**Fig. 4.5.** Catch curves for (a) P. leopardus, (b) P. maculatus and (c) P. areolatus from the ETS. The slopes of the regressions are an estimation of the rate of total mortality ($Z$) for each species.
4.3.6. Spawning seasonality

The monthly trends in GSI and mature ovarian and testicular stages indicated that the peak spawning months differed among coral trout species. Mean monthly GSI values for mature females were generally highest in October and November for *P. leopardus*, September and October for *P. maculatus* and July and August for *P. areolatus* (Fig. 4.6). The mean monthly GSI values for males were more variable than for females, but were generally highest in October and November for *P. leopardus* and September of 2004 and July 2005 for *P. maculatus* (Fig. 4.7). There were no clear peaks in monthly GSI values for *P. areolatus* males (Fig. 4.7). Missing data from the months of December, January and February precluded the specific verification of whether spawning occurred in these months for any species.

Fig. 4.6. Monthly frequencies of mature ovarian stages (bars) and mean monthly gonadosomatic index (GSI) values (lines) for female (a) *P. leopardus*, (b) *P. maculatus* and (c) *P. areolatus* from the ETS. Error bars are standard errors.
The monthly trend in the frequency of mature ovarian stages (ovaries in active vitellogenesis) suggested a similar temporal pattern in spawning activity to the monthly GSI values for all species (Fig. 4.6). The majority of spawning activity, indicated by the presence of hydrated females, occurred in October and November for *P. leopardus*, October for *P. maculatus* and July and August for *P. areolatus* (Fig. 4.6). However, there was some evidence of *P. leopardus* spawning as early as July and *P. areolatus* spawning as late as October and November (Fig. 4.6). The monthly trend in the frequency of testicular stages indicated that males of all species, particularly *P. leopardus* and *P. maculatus*, were capable of spawning over an extended period of time either side of the spawning season (as defined by the presence of active females) (Fig. 4.7). For *P. leopardus* and *P. maculatus*, 50% or more males had sperm present in all months that were sampled except *P. leopardus* in November 2005 (Fig. 4.7). For *P. areolatus*, 50% or more males had sperm present in May, July, September and November 2004 and June, July, August and October 2005 (Fig. 4.7).

Fig. 4.7. Monthly frequencies of testicular stages (bars) and mean monthly gonadosomatic index (GSI) values (lines) for male (a) *P. leopardus*, (b) *P. maculatus* and (c) *P. areolatus* from the ETS. Error bars are standard errors.
4.3.7. Length and age at sex change

There was a large overlap in the length of males and females for all species, although females generally dominated the smaller length classes and males dominated the larger length classes, particularly for \( P. \text{areolatus} \) (Fig. 4.8). Likelihood ratio tests indicated that the pattern in the length at sex change differed significantly among all species \((\chi^2 = 16.16, p = 0.003)\). Pairwise comparisons revealed that the pattern of length at sex change varied significantly between each pair of species \((P. \text{leopardus} \text{ vs. } P. \text{maculatus}: \chi^2 = 6.82, p = 0.03; \ P. \text{leopardus} \text{ vs. } P. \text{areolatus}: 24.46, p < 0.001; P. \text{maculatus} \text{ vs. } P. \text{areolatus}: \chi^2 = 24.27, p < 0.001)\). The length at which 50% of individuals changed sex was greatest for \( P. \text{areolatus} \), lowest for \( P. \text{maculatus} \) and intermediate for \( P. \text{leopardus} \) (Table 4.2). The greater overlap in the proportion of males and females in each length class and the greater length range of transitional and bisexual \( P. \text{leopardus} \) and \( P. \text{maculatus} \) indicates that these species change sex over a wider length range than \( P. \text{areolatus} \) (Fig. 4.8).

**Fig. 4.8.** Proportion of male, bisexual, transitional, and mature and immature females in each size and age class and estimated logistic function fitted to the proportion of males (relative to females) for (a) \( P. \text{leopardus} \), (b) \( P. \text{maculatus} \) and (c) \( P. \text{areolatus} \) from the ETS. Parameters of the logistic functions are given in Table 4.2.
Table 4.2. Peak spawning months and parameters of age and length-specific sex change for three species of coral trout from the ETS. $L_{50}$ and $L_{95}$ are the length (FL) at which 50% and 95% of the population are males respectively. $A_{50}$ and $A_{95}$ are the age (years) at which 50% and 95% of the population are males respectively.

<table>
<thead>
<tr>
<th>Species</th>
<th>Peak spawning months</th>
<th>$L_{50}$ (mm FL)</th>
<th>$L_{95}$ (mm FL)</th>
<th>$A_{50}$ (yrs)</th>
<th>$A_{95}$ (yrs)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>P. leopardus</em></td>
<td>Oct-Nov</td>
<td>482</td>
<td>714</td>
<td>7.2</td>
<td>14.1</td>
</tr>
<tr>
<td><em>P. maculatus</em></td>
<td>Sept-Oct</td>
<td>444</td>
<td>959</td>
<td>6.3</td>
<td>17.1</td>
</tr>
<tr>
<td><em>P. areolatus</em></td>
<td>Jul-Aug</td>
<td>549</td>
<td>617</td>
<td>9.0</td>
<td>13.8</td>
</tr>
</tbody>
</table>

The pattern of age at sex change was similar to the pattern of length at sex change for all species (Fig. 4.8). Likelihood ratio tests indicated that the pattern in the age at sex change differed significantly among all species ($\chi^2 = 13.76$, $p = 0.008$). The pattern of age at sex change for *P. areolatus* was significantly different to *P. leopardus* ($\chi^2 = 15.76$, $p < 0.001$) and *P. maculatus* ($\chi^2 = 8.22$, $p = 0.02$). The pattern of age at sex change did not differ significantly between *P. leopardus* and *P. maculatus* ($\chi^2 = 3.89$, $p = 0.14$). The age at which 50% of individuals changed sex was older for *P. areolatus* than for *P. leopardus* or *P. maculatus* (Table 4.2). Unlike the pattern of length at sex change, the age range of transitionals and bisexuals was similar among species (Fig. 4.8).

Immature *P. areolatus* were present in a number of length classes up to 520 mm FL and age classes 1 to 5 years, whereas only 2 immature *P. maculatus* and no immature *P. leopardus* were sampled (Fig. 4.8). Although there were insufficient samples to accurately estimate the length and age at maturity, our results suggest that *P. areolatus* matures at a larger size and older age than *P. leopardus* or *P. maculatus*.

The sex ratio for all species was female biased. The female:male sex ratio was most female biased for *P. areolatus* (5.41:1), followed by *P. leopardus* (2.04:1) and *P. maculatus* (1.12:1). A chi-square contingency test indicated that the sex ratios were significantly different between species ($\chi^2 = 9.60$, $p = 0.002$). The sex ratios were significantly different from 1:1 for *P. leopardus* ($\chi^2 = 49.37$, $p < 0.0001$) and *P. areolatus* ($\chi^2 = 118.34$, $p < 0.0001$) but did not differ significantly from 1:1 for *P. maculatus* ($\chi^2 = 0.68$, $p = 0.41$).

4.4. Discussion

The spatial distribution and population biology differed significantly among coral trout species in the ETS. In particular, the population biology of *P. areolatus* differed substantially to that of *P. leopardus* and *P. maculatus*. The length-weight relationship, length and age distributions, growth, peak spawning season and length and age at sex change of *P. areolatus* differed significantly to *P. leopardus* and *P. maculatus*. Age distributions and size at sex change were the only two parameters that differed significantly between *P. leopardus* and *P. maculatus*. These differences suggest that *P. areolatus* may respond differently to fishing pressure than *P. leopardus* and *P. maculatus* and, therefore, may require separate management arrangements.
The variation in the distribution of coral trout species in the ETS may reflect species-specific habitat preferences. In the ETS, *P. leopardus* were ubiquitous, suggesting that all reef habitats in the region are suitable for this species. *P. maculatus* and *P. areolatus*, however, were more confined to the western reefs, particularly in the north. These reefs are generally more sheltered from the persistent south-east trade winds and waters around these reefs are typically more turbid than reefs further east (Hemer et al. 2004, Saint-Cast and Condie 2006). *P. laevis* were only sampled from reefs in the east, and reports from fishers indicate that this species is most common even further east on the outer barrier reefs, which were not sampled in our study. Similar species-specific distributions of coral trout are found on the GBR where *P. maculatus* are abundant on island fringing reefs and inshore reefs close to the coast, *P. leopardus* are most abundant on mid-shelf reefs but are also commonly encountered on inshore and outer-shelf reefs, and *P. laevis* are most abundant on outer shelf reefs, uncommon on mid shelf reefs and rare on inshore reefs (Aying and Ayling 1983, Newman et al. 1997). *P. areolatus* are not abundant anywhere on the GBR and their distribution elsewhere is also largely unknown. The distribution of harvested coral trout from Islander and non-indigenous commercial fishers suggests that relative fishing pressure may be greater on populations of *P. areolatus* and *P. maculatus* than on *P. leopardus* and *P. laevis*, as generally higher annual catches of coral trout are taken from areas where *P. leopardus*, *P. maculatus* and *P. areolatus* are found together than from reefs where only *P. leopardus* were sampled or from outer barrier reefs where *P. laevis* is reported to be most abundant (see Chapter 3).

Maximum ages and lengths were similar among coral trout species and were similar to reported values for *P. leopardus* (Ferreira and Russ 1994, 1995) and *P. maculatus* (Ferreira and Russ 1992) from the GBR. There are no reports on the maximum age of *P. areolatus* from any location, but Pet et al. (2005) reported a similar maximum size from Indonesia. Although the maximum age and length were similar among species, the age and length frequency distributions varied significantly among species. Generally, there were older and larger *P. areolatus* than *P. leopardus* or *P. maculatus*. This may be a result of the selectivity of the sampling gear removing proportionally more older and larger *P. areolatus*, the existence of a preferred juvenile or sub-adult *P. areolatus* habitat located outside the areas sampled or poor recruitment in recent years. Interestingly, Johannes et al. (1999) found a similarly skewed length distribution of *P. areolatus* from underwater visual surveys in Palau. These authors suggested either selective fishing pressure on smaller individuals or poor recruitment in years preceding their survey may be responsible for the lack of smaller individuals in the population.

The growth of *P. areolatus* in terms of weight-at-length and length-at-age differed significantly from *P. leopardus* and *P. maculatus*. *P. areolatus* reached a larger average maximum length and was heavier for a given length than *P. leopardus* or *P. maculatus*. It should be noted, however, that FL is equivalent to TL for *P. areolatus*, as the tail for this species is convex. In contrast, the tail of *P. leopardus* and *P. maculatus* is concave. Consequently, the difference in length-weight relationships between species would be larger if total length, rather than fork length, is used. Although differences in growth between species were not large, the observed variation may have implications for reproductive output. Fecundity of individuals typically increases with body size, as the available space to store gonads also increases (Roff 1992). Therefore, fecundity of *P. areolatus* at a given size may be greater than *P. leopardus* and *P. maculatus*. Unfortunately, there are no fecundity estimates available for *P. areolatus* to test this hypothesis directly.

Some generalisations about growth of coral trout in the ETS can be drawn, despite variation in growth among species. There was a high degree of variability in size-at-
age, particularly for *P. leopardus* and *P. maculatus*, which is typical of most coral reef fish and fish populations in general. As a result, a number of age classes accumulate within a narrow size range, suggesting the use of length frequency analyses for coral trout is of little value for estimating growth. Previous studies have demonstrated an asymptotic growth pattern for *P. leopardus* (Ferreira and Russ 1994) and *P. maculatus* (Ferreira and Russ 1992) on the GBR. The growth curves for these species in the ETS were generally not of asymptotic form, due to the absence of smaller and younger fish in the samples. Ferreira and Russ (1994) demonstrated that excluding fish <350 mm TL resulted in an overestimate of $L_\infty$ and $t_0$ and an underestimate of $K$. Consequently, the VBGF parameter estimates for coral trout in our study are most likely biased. This notion is supported by the significantly lower $L_\infty$ and $t_0$ and significantly higher $K$ estimated for *P. leopardus* (Ferreira and Russ 1994) and *P. maculatus* (Ferreira and Russ 1992) using a wider length and age range.

Mortality rates did not differ significantly among coral trout species, which may be expected given the similarity in longevity among species. The estimate of $Z$ for *P. areolatus* (0.40) in our study is the first obtained for this species. Estimates of $Z$ for *P. leopardus* (0.44) and *P. maculatus* (0.61) were within the range of $Z$ estimates for these species from the GBR. Estimates of $Z$ for *P. leopardus* from the GBR have ranged from 0.12 to 0.68 (Brown et al. 1994, Russ et al. 1998). This variation most likely reflects the difficulty in estimating mortality rates using catch curves and the sensitivity of catch curves to violations of the underlying assumptions of constant recruitment and constant mortality among cohorts (Vetter 1988). Russ et al. (1998) used cohort-specific catch curves, which circumvent these assumptions, to estimate a mortality rate of 0.15 for *P. leopardus* on the GBR. This estimate may be an approximate for natural mortality for the species, as the reefs sampled by Russ et al. (1998) had been closed to fishing for several years. Ferreira and Russ (1992) estimated $Z$ for *P. maculatus* to be 0.57, which is the only other estimate available for this species.

The most important and substantial differences among coral trout species were observed in the reproductive biology. *P. areolatus* was found to spawn mainly in winter, but some spawning was observed as late as spring. In contrast, *P. leopardus* and *P. maculatus* mainly spawned in spring. The peak spawning months for *P. leopardus* (Sep-Nov) and *P. maculatus* (Sep-Oct) in the ETS are very similar to the spawning periods identified for these species on the GBR. For example, September through November was identified as the peak spawning season for *P. leopardus* in the northern (Samoilys and Squire 1994, Ferreira 1995) and central (Ferreira 1995) regions of the GBR. Similarly, Ferreira (1993) identified September through November as the peak spawning months for *P. maculatus* on the GBR. This suggests that *P. leopardus* and *P. maculatus* have a common spawning period that is similar throughout their distribution along the GBR and into Torres Strait. There are no reports of the spawning season for *P. areolatus* from the southern hemisphere, but Johannes et al. (1999) observed peaks in aggregations of *P. areolatus*, presumably related to spawning, in Palau between February and August (late winter to end summer). Although more protracted, this period encompasses the season of spawning activity identified for *P. areolatus* in the ETS. It remains unclear, however, whether spawning of any coral trout species in Torres Strait occurs between December and February, as no samples of any species were collected during these months.

The age and length at sex change for coral trout appears very flexible, as indicated by the large overlap in the age and length of males and females for all species, particularly *P. leopardus* and *P. maculatus*. Such a large overlap is indicative of a social control of sex change, whereby females are able to change sex to male to
Population Biology of Coral Trout

maintain some threshold sex ratio (Sadovy and Shapiro 1987, Cowen 1990, Vincent and Sadovy 1998) rather than changing sex at a predetermined length or age (Ghiselin 1969). However, the controlling mechanism for sex change remains unclear for the majority of protogynous fish (Sadovy 1996), including Plectropomus spp. The age and length at 50% sex change was significantly older and larger for P. areolatus than P. leopardus and P. maculatus. Consequently, the sex ratio for P. areolatus was substantially more female biased than P. leopardus and P. maculatus. Female biased populations are typical for protogynous species (Sadovy and Shapiro 1987, Sadovy 1996), where often a single male pair-spawns consecutively with a number of females (polygyny). The reason why P. areolatus was more female biased than other coral trout species in the ETS is unclear, but it may be related to spawning behaviour. Johannes et al. (1999) described the presence of all-female schools of P. areolatus passing repeatedly across aggregations of males during the spawning season in Palau. This phenomenon has not been observed for either P. leopardus or P. maculatus, which typically form aggregations containing both sexes during spawning (e.g. Samoilys and Squire 1994) and both species have been observed spawning in the same aggregations (Frisch and van Herwerden 2006). An alternate, and perhaps more plausible explanation for the heavily female biased population of P. areolatus, is that this species is less flexible in the length and age at sex change than either P. leopardus or P. maculatus. This hypothesis is supported by the narrower overlap in the size and age of P. areolatus compared with P. leopardus or P. maculatus. Adams (2003) and Davies et al. (2006) described a similar pattern for P. laevis on the GBR, which showed a highly female biased sex ratio (10:1) and a minimal overlap in sex specific distributions.

4.4.1. Implications for fisheries management

Currently, all coral trout species in the ETS are managed as a single species group, with a common legal minimum size of 380 mm TL (equivalent to 360 mm FL for P. leopardus and P. maculatus). Results from our study, however, indicate that the population biology of P. areolatus differs significantly from P. leopardus and P. maculatus, suggesting species-specific management arrangements may be required.

The current minimum legal size allows most female P. leopardus, P. maculatus and P. areolatus to spawn at least once before becoming vulnerable to harvest, which is often a primary objective of minimum size limits. However, for hermaphroditic species, it is important to account for sex change when setting size limits to avoid the disproportionate removal of one sex. In the ETS, a significant proportion of male P. leopardus and P. maculatus will be protected by the current minimum size limit, but very few, if any, male P. areolatus will be protected. Consequently, P. areolatus may be vulnerable to sperm limitation if males are disproportionately removed by fishing, as the population may be unable to compensate for male removal by earlier sex change (Vincent and Sadovy 1998). Alonzo and Mangel (2004) used a simulation model to demonstrate that populations with a less flexible size at sex change are likely to be particularly vulnerable to fishing pressure and that these populations are likely to crash if all male size classes are fished. These results strongly suggest that either the current minimum size limit for P. areolatus needs to be increased or that a maximum size limit should be introduced to protect males. The introduction of a maximum size limit for P. areolatus will need to consider the potential difficulties in enforcing such arrangements, as currently large coral trout of all species are filleted and sold together as a single product, making it difficult to trace different species.

Closures to fishing during the spawning season of coral trout is currently under consideration by fisheries stakeholders in the ETS RLF. Such closures would need to consider the variation in spawning season among coral trout species. To be effective, the closures would need to encompass at least some months of spawning for each
species, which would result in an extended closure from approximately July to October. A long closure is unlikely to be accepted by fishers, particularly non-indigenous fishers, as their fishing season occurs from February to November (see Chapter 3) and they travel long distances to the fishing grounds where they remain for the whole season (see Chapter 3). However, for spawning closures to be most effective (i.e., more effective than closures at other times), there must be evidence of intentional targeting of spawning aggregations or an increase in catchability of a species during the spawning season. There is no evidence consistent with either of these scenarios for coral trout in the ETS RLF (see Chapter 3). Therefore, spawning closures may not be the most effective tool to manage the ETS RLF.

Our study provided the first estimates of a number of demographic parameters for *P. areolatus* and demonstrated that the population biology of this species differs substantially to that of *P. leopardus* and *P. maculatus* in the ETS. Results suggest that while common management arrangements for *P. leopardus* and *P. maculatus* appear to be adequate for these species, separate management arrangements are needed for *P. areolatus* populations in the region. The current minimum legal size for coral trout in the ETS is inadequate for *P. areolatus* and an increase in the minimum legal size or the introduction of a maximum size is needed to ensure adequate protection is given to males. Additional management strategies such as closures during the spawning season are unlikely to be effective for the ETS RLF and are not likely to be accepted well by fishers.
5. Population biology of barramundi cod *Cromileptes altivelis* in the eastern Torres Strait: Implications for fishery management

5.1. Introduction

International trade in live-caught reef organisms for the live fish food trade originated in Southeast Asia and has expanded significantly since the late 1980’s throughout the Indo-Pacific region (Bentley 1999, Sadovy et al. 2003). In recent years, total trade of live reef fish into Hong Kong has been estimated to be as high as 30,000 tonnes per year (Sadovy et al. 2003). A number of commercially important finfish are heavily targeted for the live fish trade including coral trout (*Plectropomus* spp.), cods (*Epinephelus* spp.) and barramundi cod (*Cromileptes altivelis*). *C. altivelis* has been identified as one of the top three most favoured fish in the Asian restaurant industry (Watson et al. 1999, Morris et al. 2000, Pomeroy et al. 2006), fetching retail prices of up to $130USD per kilogram (Sadovy et al. 2003). The sustainability of harvests of *C. altivelis* in Southeast Asian waters is uncertain, however, as it is now uncommon to observe substantial numbers, if any, of this species in Indonesia and Malaysia (Oakley et al. 1999, Sadovy et al. 2003, Pet-Soede et al. 2004).

Due to the high economic importance of *C. altivelis* to the live fish food trade, research attention has been focussed on aquaculture and hatchery rearing of this species. Research on the biology and life history of wild populations of *C. altivelis*, however, is rare. The limited available research suggests that this species is naturally rare in abundance, has a limited range (including large regions of severely degraded reef in Southeast Asia), a relatively slow growth rate and is relatively long-lived (Oakley et al. 1999, Morris et al. 2000, Sabetian 2003, Teh et al. 2005). The most detailed description of the population biology of *C. altivelis* was provided by Davies et al. (2006), who examined populations of this species on the Great Barrier Reef (GBR). Davies et al. (2006) provided estimates of growth and mortality rates, longevity, age and length at maturity and sex change for *C. altivelis* on the GBR and demonstrated that this species has a lifespan of less than 20 years, grows quickly, and has protogynous hermaphrodite development.

It is currently unknown whether the harvest of *C. altivelis* in any fishery is sustainable due to the lack of biological information available. This has lead to the classification of this species on the IUCN Red List as ‘data deficient’. Species that have a relatively low natural abundance and have a high economic value may be more susceptible to overfishing than highly abundant species with lower economic value (Davies et al. 2006). It is essential, therefore, to obtain more accurate information on the biological parameters for *C. altivelis* to be able to assess their potential vulnerability and resilience to fishing.

In Torres Strait, *C. altivelis* are harvested as part of the multi-species RLF by both the Islander and non-indigenous commercial fisheries. A moratorium on the sale of live fish exists in the Torres Strait (Mapstone et al. 2003), but *C. altivelis* are still taken and marketed as whole gut and gilled product, and a live fish fishery in Torres Strait could still develop in the near future. Although *C. altivelis* comprise between 1-2% of the commercial harvest by numbers in the Torres Strait (Chapter 3), it is uncertain whether the current rate of harvest is sustainable. It is also unclear whether the population parameters of *C. altivelis* from the GBR estimated by Davies et al. (2006) apply to populations in the ETS. Consequently, knowledge of *C. altivelis* population dynamics in the ETS is required to evaluate the sustainability of their harvest in the
ETS RLF. The primary aim of this chapter was to provide biological data on *C. altivelis* from the ETS including estimates of the length-weight relationship, length and age distributions, growth, mortality rates, spawning season, length and age at maturity, and length and age at sex change.

### 5.2. Methods

#### 5.2.1. Sampling Methods

Biological samples of *Cromileptes altivelis* were collected during observer surveys on board Islander and non-indigenous commercial vessels. Details of observer surveys are outlined in Chapter 2. Total length (TL) and whole wet weight (W) were recorded and otoliths and gonads removed from all available samples on board non-indigenous commercial vessels.

From March 2004 to November 2005, samples of whole frozen *C. altivelis* were purchased monthly from Islander and non-indigenous commercial operators in the ETS, including those samples under the 450 mm minimum size limit from non-indigenous observer surveys. It was necessary to purchase whole fish as *C. altivelis* are highly valued and marketed as gutted and gilled whole fish. Samples were not available from all months in each year due to the opportunistic nature of sample collection from commercial fishers. The TL was measured and otoliths and gonads removed from each individual. Sex information was not available from all samples, as some fish had been gutted and the gonads were missing.

#### 5.2.2. Sample Processing

All otoliths were processed using the techniques outlined in Chapter 4. Otoliths were read twice by a single experienced reader. If the counts from the two readings agreed, the age was accepted as the final age. If the counts differed, a third count was made by the same reader. If the count from the third read agreed with the count from either the first or second read, that count was accepted. If there was no agreement among all reads (44 of 188 otoliths), the median count of three reads was accepted as the final age.

Gonads were frozen immediately after removal aboard commercial vessels and transported back to the laboratory. Gonads from all samples were thawed and stored in a preservative solution of 10% phosphate buffered formalin. All gonads were weighed, processed and classified into reproductive stages according to the procedures outlined in Chapter 4. Gonad weight was estimated for 16 samples from which only one lobe was available.

#### 5.2.3. Biological parameters

Estimates of the length weight relationship, length and age distributions, growth, mortality rate, spawning season, length and age at sex change were made for *C. altivelis* using the same methods as for the three coral trout species outlined in Chapter 4. Reliable estimates of the length and age at maturity for *C. altivelis* were not possible due to the absence of immature individuals from all samples.

### 5.3. Results

#### 5.3.1. Age, Growth and Mortality

A total of 190 samples of *C. altivelis* were collected, but the same information was not available for all fish. Weight information was available for only 30 individuals ranging in TL from 425 mm to 620 mm (Fig. 5.1). The relationship between TL and W was approximately isometric ($b \approx 3$).
Length and age information was available from 186 individuals ranging in size from 381-669 mm TL and in age from 3-16 years (Fig. 5.2). The majority of samples were between 440-610 mm TL, with a mode at 490 mm TL (Fig. 5.2a). Most fish were between 5-12 years old, with a mode at 7 years (Fig. 5.2b). These distributions indicate that *C. altivelis* are not fully selected by the sampling gear until 490 mm TL and 7 years of age. These results comprise both purchased samples from the fleet and observer survey samples, with the non-Indigenous observer samples unconstrained by the minimum size limit of 450 mm TL.

![Graph](image)

**Fig. 5.1.** Weight at length data and fitted power curve for *C. altivelis* from the ETS.

![Graph](image)

**Fig. 5.2.** Length (a) and age (b) frequency distributions for *C. altivelis* from the ETS.

There was substantial variation in length-at-age for *C. altivelis* from the ETS. The growth curve fitted to the length-at-age data was relatively flat as few fish smaller than 400 mm FL and less than 3 years of age were sampled due to the selectivity of the fishing gear (Fig. 5.3). Parameter estimates for the fitted von Bertalanffy growth function (VBGF, see Chapter 4) for *C. altivelis* in the ETS are as follows: $L_\infty = 623$ mm, $K = 0.13$ and $t_0 = -6.67$ years. The growth curve was not generally of asymptotic...
form and estimates of VBGF parameters were most likely biased due to the absence of small and young fish. This growth pattern is, however, indicative of fish selected by the gear, and suggests that juveniles grow rapidly in their first few years of life to reach over 400 mm $TL$, followed by a period of relatively slower growth.

![Graph showing total length vs. age](image)

**Fig. 5.3.** Length-at-age data and fitted VBGF curve for *C. altivelis* from the ETS.

The rate of total mortality ($Z$) for *C. altivelis* was estimated using data from age classes 7 to 16 (Fig. 5.4) under the assumption that fish are fully recruited to the sampling gear by this age. The linear regression provided a reasonably good fit to the data giving a $Z$ estimate of 0.28, corresponding to an annual survival rate of 75%.

![Graph showing catch curve](image)

**Fig. 5.4.** Catch curve for *C. altivelis* from the ETS. The slope of the regression is an estimation of the rate of total mortality ($Z$).

### 5.3.2. Reproductive biology

There were no immature *C. altivelis* identified in the sample. Hence it was not possible to estimate size or age at maturity statistically. However, the smallest and youngest (mature) female sampled was 381 mm TL and 3 years of age, indicating that first maturity is likely to occur at a size and age no larger or older than this.
**Spawning seasonality**

As samples were not collected during all months of the year for the entire sampling period, missing data from these months precluded the verification of spawning in all months. Gonadosomatic Index (GSI) was therefore calculated for all samples by pooling data across years and dividing into six seasonal intervals (Fig. 5.5). Mean GSI was greatest for females in October-November (Fig. 5.5a). The seasonal pattern in mean GSI for males was similar to females, reaching a minimum in April-May, increasing through to October-November, but reached a peak in February-March (Fig. 5.5b).

![Fig. 5.5 Seasonal frequencies of mature developmental stages (bars) and mean monthly gonadosomatic index (GSI) values (lines) for female (a) and male (b) C. altivelis from the ETS. Error bars are standard errors.](image)

The monthly trend in the frequency of mature ovarian stages (ovaries in active vitellogenesis) suggested a similar temporal pattern in spawning activity to the mean monthly GSI values. The presence of hydrated females indicated the majority of spawning activity, which occurred in October-November (Fig. 5.5). Although the trend for males was less clear, the frequency of testicular stages (with ripe spermatozoa) indicated that males were capable of spawning throughout the year.
**Length and age at sex change**

There was a large overlap in the length range of males and females (421-621 mm), although females generally dominated the smaller length classes and males dominated the larger length classes (Fig. 5.6a). The length at which 50% of individuals changed sex from female to male, estimated from the logistic function, was 564 mm (Fig. 5.6a).

![Graph showing proportion of females, transitionals, and males in each size and age class](image)

**Fig. 5.6.** Proportion of females, transitionals and males in each size (a) and age (b) class and estimated logistic function fitted to the proportion of males (relative to females) for *C. altivelis* from the ETS.

A similar trend was observed for age at sex change, with an overlap of females and males across most age classes (3-13 years, with one female individual at 16 years, Fig. 4.6b). Females dominated the younger age classes and the age at which 50% of the individuals changed sex to male, estimated from the logistic function, was 9.6 years. Only 1 transitional individual (500 mm TL, no age estimate) was identified, which represents approximately 0.5% of the total sample.

The sex ratio of *C. altivelis* was female biased (1.4:1), with 103 females and 73 males in the sample. The sex ratio differed significantly from 1:1 ($\chi^2 = 5.11, p = 0.02$).

**5.4. Discussion**

This study provides some of the only estimates of biological parameters for *C. altivelis* and the first estimates for populations in Torres Strait. Populations of *C. altivelis* in Torres Strait were found to exhibit a moderate longevity (16 years), with a relatively fast growth rate in the first few years of life, followed by a period of relatively slow growth. *C. altivelis* were also confirmed to be protogynous hermaphrodites in Torres Strait.

The population biology of *C. altivelis* in Torres Strait is very similar to populations on the GBR. For example, *C. altivelis* populations on the GBR displayed a similar pattern of growth and reached a maximum age of 19 years (Davies et al. 2006). The distribution of lengths and ages of *C. altivelis* populations in Torres Strait was also very similar to populations on the GBR, with both populations sharing very similar...
length and age modes. The age size at which 50% of the population changed sex from female to male was also similar between Torres Strait (9.6 years, 564 mm) and GBR (8.2 years, 542 mm) populations, which is around two thirds of their maximum age and length.

The total mortality rate estimated from catch curves for *C. altivelis* \((Z = 0.28)\) was identical to that estimated for populations on the GBR (Davies et al. 2006). It is not possible to derive a direct estimate of natural mortality for populations of *C. altivelis* in Torres Strait, as all reefs in Torres Strait have been fished for many years. However, Hoenig (1983) provides a method to derive indirect estimates of natural mortality based on longevity. Using this method and an estimate of 16 years longevity, natural mortality for *C. altivelis* in Torres Strait is estimated to be 0.28. This estimate suggests that the estimate of total mortality or longevity was underestimated in this study. The total mortality rate estimated from catch curves for *C. altivelis* is identical to that estimated for populations on the GBR (Davies et al. 2006). The total mortality rate estimated from catch curves for *C. altivelis* (\(Z = 0.28\)) was identical to that estimated for populations on the GBR (Davies et al. 2006). It is not possible to derive a direct estimate of natural mortality for populations of *C. altivelis* in Torres Strait, as all reefs in Torres Strait have been fished for many years. However, Hoenig (1983) provides a method to derive indirect estimates of natural mortality based on longevity. Using this method and an estimate of 16 years longevity, natural mortality for *C. altivelis* in Torres Strait is estimated to be 0.28. This estimate suggests that the estimate of total mortality or longevity was underestimated in this study or that fishing mortality for *C. altivelis* in Torres Strait is negligible. Fishing mortality is unlikely to be negligible in Torres Strait, as *C. altivelis* comprises over 2% of the harvest by weight (Chapter 3), which equates to approximately 3 t. It is more likely that longevity in this study was underestimated, particularly given that longevity was estimated to be 19 years for populations on the GBR (Davies et al. 2006), which was based on a larger sample size. Using the Hoenig (1983) method, natural mortality is estimated to be 0.23 for a longevity of 19 years.

The size and age distributions and the distribution of size-at-age indicate that a large proportion of the *C. altivelis* populations in Torres Strait below 430 mm TL and 7 years of age are not selected by the fishing gear used in the fishery. Consequently, estimates of growth parameters are likely to be biased, as the gear disproportionately selected for the faster growing young individuals. Although some caution should be exercised when using the growth parameters estimated in this study, the data and results of the analysis do provide useful information to better understand the biology of the *C. altivelis* and highlight important issues for future research.

Serranids were thought to typically be long-lived fishes (Manooch 1987, Sadovy et al. 2003), but recent studies have demonstrated longevity within this group varies greatly (Grandcourt 2002, Pogonoski et al. 2002, Davies et al. 2006). The maximum age of *C. altivelis* in Torres Strait (16 years) and the GBR (19 years) suggest that this species has a moderate longevity in comparison to other large serranids such as *Mycteroperca interstitialis* at 41 years (Manickchand-Heilemen and Phillip 2000) and *Epinephelus fuscoguttatus* at 42 years of age (Pears et al. 2006).

Spawning behaviour has been reported for a number of serranid species (Chapter 4, Sadovy et al. 1994, Coleman et al. 1996, Samoilys 1997, Sluka 2001b, Whaylen et al. 2004, Matos-Caraballo et al. 2006), but *C. altivelis* spawning in the wild has not been documented (Morris et al. 2000). Spawning has been observed in captivity at Indonesian aquaculture facilities, where spawning occurred throughout most of the year (Mous 2000, Sugama et al. 2003). However, these observations are likely to be very biased, as aquaculture conditions are typically manipulated to maximise the number of spawning events throughout the year. Peak periods of spawning activity of *C. altivelis* in Torres Strait (based on peak GSI and the presence of ripe females) were identified in the months of October-November. It is unclear whether the spawning period extended into December and January due to the absence of samples from these months. Male *C. altivelis* in Torres Strait were found to be ripe and capable of spawning throughout the year.

Plasticity often exists in the timing of sex change for fish populations in different regions and locations. The large overlap in the size and age of *C. altivelis* indicates that sex change is also flexible in this species. Such a large overlap is indicative of a
social control of sex change, whereby females are able to change sex to male to maintain some threshold sex ratio (Shapiro 1981, Ross et al. 1983, Sadovy and Shapiro 1987, Cowen 1990, Vincent and Sadovy 1998) rather than changing sex at a predetermined size or age. However, the controlling mechanism for sex change remains unclear for the majority of protogynous fish (Sadovy 1996), including C. altivelis. The sex ratio of populations of C. altivelis in Torres Strait was slightly female biased, which is typical for protogynous species (Sadovy and Shapiro 1987, Sadovy 1996) and suggests a polygynous mating pattern for C. altivelis, whereby a single male pair-spawns consecutively with a number of females.

5.4.5. Implications for fisheries management

Although C. altivelis populations are considered to be threatened by overfishing from the live food fish trade in waters of the Indo-Pacific (Oakley et al. 1999, Morris et al. 2000, Pogonoski et al. 2002, Sadovy et al. 2003, Teh et al. 2005) and are listed as a no-take species on the east coast of Australia, management strategies in Torres Strait provide some protection for this species. The minimum legal size limit of 450 mm TL was implemented on the basis that it would allow C. altivelis to spawn at least once before becoming vulnerable to harvest by the fishery. Given that all individuals sampled in this study were mature and the smallest individual sampled was 381 mm TL, the current minimum legal size appears to be more than adequate in achieving this objective. Furthermore, this minimum legal size allows a small proportion of individuals to change sex from female to male before becoming vulnerable to harvest, which reduces the probability of the populations becoming extremely female biased and sperm limited.

C. altivelis is not known to aggregate in particular locations to feed or spawn. Therefore, this species is unlikely to be vulnerable to specific targeting by fishers. However, the low abundance of this species elsewhere (Pogonoski et al. 2002) is cause for concern. Along its range on the GBR, catch rates indicate that C. altivelis abundance is many times lower than other serranids such as common coral trout (Davies et al. 2006). The relative abundance of C. altivelis in Torres Strait is unknown. However, it will be important to monitor the harvest levels of C. altivelis in Torres Strait to ensure that populations are being overfished.

This study provides essential information for evaluating whether the harvest of C. altivelis in the RLF of the ETS is currently managed in a sustainable manner. Generally, the estimates of biological parameters from Torres Strait were similar to those estimated for populations on the GBR by Davies et al. (2006). Therefore, given that the two regions are adjacent, combining the information from both studies provides valuable information for the management of C. altivelis populations throughout the GBR and Torres Strait.
6. Evaluation of alternative management strategies for the eastern Torres Strait commercial reef line fishery

6.1. Introduction

6.1.1 Background

Commercial harvest of reef fish in the ETS is shared between Torres Strait Islanders and non-indigenous fishers, but to date no formal assessment of the fishery has been done, and no resource allocation or management strategies directed at the special circumstances in the Torres Strait currently exist. The development of a conventional stock assessment for the ETS RLF is currently not possible, due mostly to the lack of independent abundance estimates, but also due to the spatial complexity of reefs and the resulting fragmented populations. Conventional stock assessments also are typically retrospective rather than forward-looking and reactive rather than proactive (Hilborn and Walters 1992). Therefore, management planning for the future is often based on observations from the past rather than quantitative predictions of the future.

Management Strategy Evaluation (MSE) (Smith 1993, 1994) is an alternative approach to conventional fishery assessments, with the advantage of being able to assess the medium to long-term performance of a range of management strategies and compare the likely outcomes from each. Importantly, MSE provides a system for comparing the performance of alternative management strategies against a range of different stakeholders’ objectives based on a common currency across multiple and often divergent objectives. Unlike many conventional fishery assessments, MSE does not necessarily require independent estimates of abundance, as MSE operates on a relative basis rather than within an absolute framework. Consequently, catch and effort data and biological information analysed in this report is sufficient to develop an MSE for the ETS RLF.

In a recent review of the status of the RLF in the ETS, Mapstone et al. (2003) recommended that future research should include formal assessments of the likely strengths and weaknesses of potential management strategies for the fishery. MSEs have been developed for the reef line fishery on Queensland’s east coast using the Effects of Line Fishing Simulator (ELFSim) (Mapstone et al. 2004, Little et al. 2007a, b). In this chapter, we follow this recommendation by adapting ELFSim to develop a MSE for the ETS RLF.

6.1.2. ELFSim

The Effects of Line Fishing Simulator (ELFSim, Mapstone et al. 2004, Little et al. 2007a, b) is a decision support tool designed to evaluate the options available for managing the harvest of coral trout in the reef line fishery on the GBR. ELFSim contains several components, including output visualisation and run management, but the most important components are a spatially-structured biological model of the (P. leopardus) population dynamics, and a model of fishing behaviour (Fig. 6.1). ELFSim operates with a monthly time step, and each simulation consists of two parts (Fig. 6.1). The first (‘initialization’) step operates historically from 1965 to the ‘present’ (in this case 2004), using information from visual surveys, catch records from the fishery, and the physical characteristics of the reefs to determine the size of the population on each reef at present. The reef populations are then projected into the future subjecting them to simulated fishing pressure, which is, in turn, affected by user-specified management regulations. The management regulations available involve area closures (including rotational closures), changes to gear selectivity,
minimum legal sizes, as well as annual allowable effort for each of the fishing sectors. More recently, management of the GBR fishery has moved to an Individual Transferable Quota (ITQ) system placed on the commercial sector and the model is being further developed to implement ITQs as well as effort controls on the harvest component. The consequences of various alternative combinations of management regulations (specifications for effort levels, spatial closures, gear selectivity, etc.) are summarised in terms of biological and fishery performance indicators during the projection period. ELFSim can incorporate multiple ‘vessel-classes’, each of which potentially uses a different strategy for allocating effort spatially and temporally. The current implementation includes three such classes to represent the commercial, recreational and charter sectors of the reef line fishery on the GBR.

Fig. 6.1. Flowchart of main steps in ELFSim (Source: Little et al. 2007b)
ELFSim provides an MSE framework to examine tradeoffs associated with the performance of alternative conservation and fishery management strategies, which to date have focused on area closures and effort controls respectively. ELFSim is comprised of three integrated components: 1) biological operating model; 2) effort model; and 3) management model. The biological component incorporates the underlying population dynamics of the target species, including vital life history characteristics such as growth, reproduction and mortality. The harvest component incorporates the fishery dynamics and subsequent harvest of the target populations represented in the biological model. The management component allows various management strategies, including spatial and temporal closures, size limits and effort restrictions, to be imposed. ELFSim also incorporates uncertainty in the fish population and fishery dynamics, including variability in recruitment and catchability (Mapstone et al. 2004, Little et al. 2007a, b).

6.1.3. Objectives
Understanding fishery systems should involve an approach where not only knowledge of the biology of the targeted species but also fisher behaviour is required. In this chapter, we build on previous related research with the objectives to:

1. Apply the ELFSim model to the catch and effort characteristics of the non-indigenous and Islander commercial sectors of the ETS RLF, and

2. formally evaluate alternative harvest and conservation management strategies for coral trout (all species combined) for the ETS RLF.

The work consisted of two components. In the first component, the spatial extent of ELFSim was modified to capture the reefs in the ETS and the characteristics of these reefs, including location and reef perimeter.

The second component of the project used the ELFSim model tailored to the ETS to evaluate stakeholder-derived management strategies for coral trout (all species combined). Specific operational management objectives, performance measures and management strategies were defined for coral trout in consultation with diverse stakeholders and formally evaluated using ELFSim.

6.2. Methods

6.2.1. Fishery data used in ELFSim
Annual catches of coral trout in Torres Strait were derived from the non-indigenous and Islander sectors to construct a time series of total catch and effort for input into ELFSim from 1965-2004 (Fig. 6.2). Actual catch and effort data exist from 1988 for both sectors. For the years prior to these dates, catches were interpolated linearly for each calendar month backwards to a value of zero in 1965, when it was assumed the fishery started. The starting point for these interpolations was 1987, the year immediately prior to the earliest year for which actual data exist. At this point, the catch for each month was assigned the average of reported values in that month from 1988-2004 for each sector. This strategy preserves the historical seasonality dynamics observed in the fishery since 1988. The spatial distributions of catch and effort for each sector were estimated at the scale of individual reefs using the procedure outlined in Chapter 3.
6.2.2. Biological model

An integral component of ELFSim is the biological model which captures the underlying population dynamics of the target species and includes growth, reproduction and mortality. The biological model for coral trout (Mapstone et al. 2004, Little et al. 2007a, b) was based on earlier models developed by Walters and Sainsbury (1990) and Mapstone et al. (1996). The target species are assumed to exist as widespread metapopulations consisting of relatively discrete post-settlement subpopulations, each associated with an individual reef. The population dynamics model is age-, sex- and size-structured, assumes that the number of 0-year-olds is related to the size of the reproductive component of the population according to a stock-recruitment relationship, takes account of sex change and allows for larval migration among reefs and variation in settlement (Punt et al. 2001, Mapstone et al. 2004).

Several sources of process error (Francis and Shotton 1997) such as variation in natural mortality and larval survival are included in the model. The model considers fish populations on individual reefs and accounts for the effects of the catches and effort reported by sectors of the fishery. The modular nature of ELFSim readily facilitates the extension of the single-species population dynamics model developed for coral trout (Mapstone et al. 2004) to a two-species (or more) population dynamics model incorporating other potential target species. In this study, however, the biological model included a single species representing a combination of common coral trout (P. leopardus), passionfruit trout (P. areolatus), bar cheeked trout (P. maculatus) and blue spot trout (P. laevis). These species form the known coral trout species complex in Torres Strait and comprise the majority of the catch by non-indigenous and Islander commercial fishers (see Chapter 3). It was not possible to
develop a separate biological model for each coral trout species, as all commercial catches are reported as a single species group (coral trout) and there is no way to separate catches into individual species.

Although biological parameters were estimated for *P. leopardus*, *P. areolatus* and *P. maculatus* (Chapter 4), we used biological parameters for common coral trout from the GBR to parameterise the biological model. This was because *P. leopardus* is the most abundant species in the commercial catch in Torres Strait (Chapter 3) and because the biological parameters estimated for Torres Strait populations were biased due the absence of smaller and younger individuals in samples collected.

The model allows for movement of coral trout larvae among reefs but ignores the possibility of movement of coral trout aged 1 year and older (see Davies 1995).

**Population dynamics model**

For ease of presentation, the equations below assume that the parameters determining fecundity, sex-change and growth are independent of reef, although the software that implements the model has the functionality to allow these parameters to be specified as reef dependent.

**Basic population dynamics**

The basic population dynamics are defined by the equations:

$$
N_{r,y+1,a} = \begin{cases} 
N_{r,y+1,a} & a = 0 \\
\sum_{r'} N_{r',y+1,a} & a = 1 \\
\sum_{r'} N_{r',y+1,2,a-1} e^{-Z_{r',y+1,2,a-1}} & a = 2, \ldots, x-1 \\
(\sum_{r'} (N_{r',y+1,2,x-1} e^{-Z_{r',y+1,2,x-1}} + N_{r',y+1,2,x} e^{-Z_{r',y+1,2,x}})) & a = x 
\end{cases}
$$

(6.1)

where $N_{r,y,a}$ is the number of fish of age $a$ on reef $r$ at the start of year $y$,

$N_{r,y,m,a}$ is the number of fish of age $a$ on reef $r$ at the start of month $m$ of year $y$ (by definition $N_{r,y+1,0,a} = N_{r,y,a}$):

$$
N_{r,y+1,m,a} = N_{r,y,m,a} e^{-Z_{r,y+1,m,a}}
$$

(6.2)

$Z_{r,y,m,a}$ is the total mortality on fish of age $a$ on reef $r$ during month $m$ of year $y$:

$$
Z_{r,y,m,a} = M_{r,y,a} / 12 + \sum_{f} F_{r,y,m,a,f}
$$

(6.3)

$M_{r,y,a}$ is the instantaneous rate of natural mortality on fish of age $a$ during year $y$. 


$F_{y,m,a,f}^r$ is the fishing mortality on fish of age $a$ on reef $r$ during month $m$ of year $y$ by vessel-class $f$, (0: non-indigenous, 1: Islander) and $x$ is the maximum age considered (taken to be a "plus group").

The maximum age $x$ (18 years) for coral trout (Table 6.1) has little effect on the results because the rate of natural mortality (0.45 yr$^{-1}$; Table 6.1) assumed for fish aged 2 years and older implies that a relatively small number of fish reach that age.

0-Year olds
All fish are born as females, where births are assumed to occur at the start of the year following Ferreira and Russ (1994) and Russ et al. (1996). The number of 0-year-olds on reef $r$ at the start of year $y$ is determined from a contribution from spawning on reef $r$ and from a contribution from all reefs (Mapstone et al. 1996):

$$N_{y,0}^r = \left[ st \bar{f}^r S_y^r + (1-st)c^r B L_y^r \right]$$

where $S_y^r$ is size of the reproductive component of the population on reef $r$ at the start of year $y$ (taken to be the biomass of mature females – also referred to as the spawner biomass):

$$S_y^r = \sum_{a=1}^x f_{L_a} w_{L_a} N_{y,a}^r (1 - P_{L_a})$$

$$\bar{f}^r = \frac{N_{y,0}^r}{S_0^r}; \ c^r = \frac{\sum_r N_{y,0}^r}{\sum_r N_{y,0}^r}$$

$S_0^r$ is the size of the reproductive component of the population on reef $r$ at pre-exploitation equilibrium,

$N_{y,0}^r$ is the number of 0-year-olds on reef $r$ at pre-exploitation equilibrium,

$st$ is the fraction of the larvae that settle on reef $r$ that originated from reef $r$,

$L_a$ is the length of a fish of age $a$

$w_L$ is the mass of a fish of length $L$,

$f_{L_a}$ is the proportion of animals of length $L$ that are mature,

$P_{L_a}$ is the proportion of fish of length $L$ that are male,

$B L_y^r$ is the background supply of larvae to reef $r$ from all reefs during year $y$. 
\[ BL'_y = \sum_{r'} \tilde{f}^{r'} S^{r'} \Omega^{r'r} \]  

(6.7)

\( \tilde{c}^{r'} \) is the scaling factor for reef \( r \) to account for variation in background larval supply among reefs, and 

\( \Omega^{r'r} \) is the fraction of larvae that move from reef \( r' \) to reef \( r \).

The proportion of fish that are mature, \( f_L \), and those that are male, \( P_L \), are determined from logistic functions of fish length (Table 6.2). The values in the larval dispersal matrix, \( \Omega \), are proportional to the fraction of larvae that move from reef \( r' \) to reef \( r \) because the value for \( \tilde{c}^{r'} \) provides an overall scaling factor. The values in the larval dispersal matrix are determined using a distance-based distribution of larvae: 

\[ \Omega^{r'r} = \exp(-3.4d(r,r') - 3.91) \]  

where the function \( d(r,r') \) is the distance between the centres of reefs \( r \) and \( r' \) (in degrees). This relationship is based on fitting a linear model to the logarithm of the fraction of the larvae which move from one reef to another derived from a hydrodynamic model (James et al. 2002) of larval movement for a hypothetical species over 324 reefs in the Cairns region of the GBR (Fig. 6.3).

**Fig. 6.3.** Data plot and exponential line fit of probability of larval migration between reefs as a function of distance (in degrees).
The value of $c^r$ (see Little et al. 2007a for derivation) depends on the larval dispersal matrix. The value of $c^r$ is recalculated annually for scenarios in which the larval dispersal matrix is based on the model of larval advection and behaviour, and hence varies among years.

**Recruitment to reefs**

The number of 1-year-olds on reef $r$ at the start of year $y+1$ is the number of 0-year-olds on reef $r$ the previous year modified by the density-dependent mortality between ages 0 and 1 plus the effect of random environmental variability and ‘recruitment pulses’. These stochastic or ‘noise’ terms can also be considered to effect larval mortality rather than that between larvae 0 and 1:

\[
N_{y+1,1}^r = N_{y,0}^r e^{-\beta^r U_{y+1}^r / U_0^r} e_{y}^{\epsilon_{y} - \sigma_y^2 / 2} \sum_{i} e^{\omega_p dist(r, c_i)} \exp(\epsilon_{y} dist(r, c_i))
\]  

(6.8a)

\[
U_{y+1}^r = N_{y,0}^r e^{-\beta^r U_0^r} + \sum_{a=2}^{J} N_{y+1,a}^r
\]  

(6.8b)

\[
\epsilon_{y} = \tau_r z_y + \sqrt{1 - \tau_r^2} z_y'
\]  

(6.8c)

where $\beta^r$ is the density-dependence parameter for reef $r$,

$U_0^r$ is the value of $U_y^r$ at pre-exploitation equilibrium,

$J$ is the maximum age of a ‘juvenile’,

$z_y, z_y'$ are independent and identically distributed random deviates from $N(0; \sigma_r^2)$,

$\sigma_r^2$ is the overall inter-annual variation in larval abundance,

$\tau_r$ is the correlation in larval abundance among reefs,

$x_{y,i}$ is the magnitude of the $i$th ‘recruitment pulse’ during year $y$, generated from the normal distribution, $N(0; 1^2)$,

$\omega_p$ is the parameter that determines the spatial extent of a ‘recruitment pulse’, and

$c_i$ is the centre of the $i$th ‘recruitment pulse’.

The value for the parameter $\beta^r$ is determined by solving the system of equations for a pre-specified value stock-recruitment steepness parameter, $h$ (Little et al. 2007a, b). Steepness is defined after Francis (1992) to be the fraction of the pre-exploitation number of 1-year-olds to be expected when the spawner biomass is reduced to 20% of its (average) pre-exploitation level.
Natural mortality

The model used to determine natural mortality by age and year allows for differences in the mean value of natural mortality among ages, variability in natural mortality over time, the effect of catastrophic events, and time-trends in natural mortality:

\[
M_{y,a} = (M_{y-1,a})^{\tau^M} (M_{y,a})^{1-\tau^M} e^{\left[\frac{\sigma_M^2}{2} \right] y^2 - \left(\frac{\sigma_M}{2}\right)^2 y^2} \epsilon_{y,a}^M \sim N(0; (\sigma_M^M)^2)
\]  

(6.9a)

\[
M'_{y,a} = \begin{cases} 
M_a + M_c \eta_y & \text{if } y \leq y_{fst} \\
M_a + M_{fin,a} \frac{(y-y_{fst})}{(y_{fst}-y_{fst})} + M_c \eta_y & \text{if } y_{fst} < y < y_{lst} \\
M_a + M_{fin,a} + M_c \eta_y & \text{otherwise}
\end{cases}
\]  

(6.9b)

where

- \(M_a\) is the expected rate of natural mortality on fish of age \(a\),
- \(\sigma_M\) is the parameter that determines the extent of temporal variation in natural mortality,
- \(\tau^M\) determines the extent of temporal correlation in natural mortality,
- \(y_{fst}\) is the year in which the natural mortality rate begins to change,
- \(y_{lst}\) is the year after which the natural mortality rate ceases to change and remains constant,
- \(M_{fin,a}\) is the amount by which natural mortality changes between years \(y_{fst}\) and \(y_{lst}\) for a fish of age \(a\),
- \(M_c\) is the amount by which natural mortality increases during a catastrophic event,
- \(\eta_y\) is a random variable that is 1 with probability \(p_c\) and 0 otherwise, and
- \(p_c\) is the probability of a catastrophic event.

Equation (6.9) allows for catastrophic events (such as the effect of a cyclone) to increase natural mortality on all fish by \(M_c\) yr\(^{-1}\). The probability of a catastrophic event is assumed to be \(p_c\) (base-case value zero). The value of \(\eta_y\) is independent of reef so that it is assumed a catastrophic event has the same effect across all of the reefs included in the model. More spatially restricted catastrophic events could be implemented with relative ease. Time-trends in natural mortality cause natural mortality for age \(a\) to increase from \(M_a\) to \(M_a + M_{fin,a}\) over the years \(y_{fst}\) to \(y_{lst}\). This formulation provides a framework within which some of the possible effects of global climate change can be investigated. The simulations in this report ignore the
possibility of climate change and catastrophic events. The values for the remaining parameters that determine natural mortality ($M_a$, $\sigma^M$ and $\tau^M$) are listed in Table 6.1.

Growth
The growth of an individual is assumed to follow the VBGF:

$$L_a = \ell_\infty (1 - e^{-\kappa t_{a-0}})$$

(6.10)

where $\ell_\infty$ is the maximum length

$\kappa$ is the von Bertalanffy growth rate

$t_{a0}$ is the length at age 0

The values for the parameters that determine growth (i.e., values for $\kappa$, $\ell_\infty$, and $t_{a0}$) are determined by fitting a model to data collected on length-at-age, after accounting for gear selectivity (Mapstone et al. 2004).

Mass as a function of length is determined using the standard allometric equation:

$$w_L = b_1 L^{b_2}$$

(6.11)

where $b_1, b_2$ are the parameters of the relationship between length and mass (Table 6.1).

6.2.3. Catches
The landed catch (in mass, kg) of fish from reef $r$ during month $m$ of year $y$ by vessel-class $f$, $C_{y,m,f}^r$, is computed using the equation:

$$C_{y,m,f}^r = \sum_{a=0}^{k} \sum_{0}^{D'_r} \frac{D'_a}{Q_{y,m,a,f}} \frac{F_{y,m,a,f}^{r,k}}{Z_{y,m,a}^{r,k}} N_{y,m,a}^{r,k} (1 - e^{-Z_{y,m,a}^{r,k}})$$

(6.12a)

$$F_{y,m,a,f}^{r,k} = Q_{y,m,a,f} L_{y,m,a}^{r,k} V_{y,m,a}^{r,k} F_{y,m,f}^r$$

(6.12b)

where $Q_L = D'_L + R'_L$

$$D'_L = \begin{cases} D, & \text{for } L < L_{\text{MLS}} \\ 1, & \text{otherwise} \end{cases}$$
\[ R'_L = \begin{cases} 
R & \text{for } L < L_{\text{MLS}} \\
0 & \text{otherwise} 
\end{cases} \]

\( L_{\text{MLS}} \) is the minimum legal size,

\( D \) is the fraction of fish that are retained following capture,

\( R \) is the fraction of fish that die after being released,

\( V_L \) is the selectivity of the gear on fish of length \( L \),

\( F_{r,ym,f}^r \) is the "fully-selected" fishing mortality applied to reef \( r \) by vessel-class \( f \) during month \( m \) of year \( y \):

\[
F_{y,ym,f}^r = q'_f \left( \frac{B_{y,ym}^r}{B_0^r} \right)^\phi E_{y,ym,f}^r e^{\phi - \phi^2 / 2} \tag{6.13}
\]

\( B_{y,ym}^r \) is the biomass on reef \( r \) at the start of month \( m \) of year \( y \) available to the fishery (i.e., exploitable biomass):

\[
B_{y,ym}^r = \sum_{a=0}^{\infty} \sum_{k} W_{ka} D'_{y,ym,a}^{(a-0.5)/12} D_{y,ym,a}^{(a-0.5)/12} V_{y,ym,a}^{(a-0.5)/12} N_{y,ym,a}^{(k-0.5)/12} \tag{6.14}
\]

\( B_0^r \) is the value of \( B_{y,ym}^r \) at the pre-exploitation equilibrium level,

\( \phi \) is a parameter that permits catchability to be density-dependent,

\( E_{y,ym,f}^r \) is the effort applied by vessel-class \( f \) on reef \( r \) during month \( m \) of year \( y \),

\( \varsigma_{y,ym,f}^r \) is a factor to account for random variation in catchability \( (\varsigma_{y,ym,f}^r \sim N(0; \sigma_{\varsigma}^2)) \), and

\( q'_f \) is the catchability coefficient for vessel-class \( f \) and reef \( r \).

The catch, therefore, is a function of the biomass available to the fishery on each reef at the start of the month, amount of effort from each sector of the fishery applied to that reef in that month, selectivity function and catchability coefficient. Selectivity is assumed to be constant across sectors, and to depend on length. Retained and discarded catch are defined according to the current minimum legal size (MLS) for common coral trout which is 38 cm TL.

If catch and effort data are available for reef \( r \), the catchability coefficients for each vessel-class are computed using the formula:

\[
q'_f = \exp\left( \frac{1}{n_r n_m} \sum_{y} \sum_{m} \ln\left( \frac{F_{y,ym,f}^r}{E_{y,ym,f}^r / (B_{y,ym}^r / B_0^r)^\phi} \right) \right) \tag{6.15}
\]
where \( n_y \) and \( n_m \) are the number of years and months over which the data span. The variance is calculated as

\[
\sigma_{q_f}^2 = \frac{1}{n_y n_m} \sum_y \sum_m \left[ \ln\left( \frac{F_{y,m,f}}{E_{y,m,f}} \right) \right]^2
\]

where the summations over year are restricted to the years for which effort data are available (see below Effort Model). This approach cannot be applied to reefs for which there are no catch and effort data. Therefore, the non-indigenous catchability coefficients for the few reefs for which there are no catch and effort data are taken to be equal to the catchability coefficient of the closest reef. This approach is not used for the Islander sector because there are many reefs without Islander catch and effort data and so the approach would lead to an unrealistically exaggerated spread of Islander effort across the Torres Strait in the future. Instead, the catchability coefficients for the Islander vessel-class for reefs without catch and effort information is set equal to zero which, given the effort allocation algorithm, prevents effort occurring on those reefs in the future.

### 6.2.4. Initial conditions

The population is assumed to have been at pre-exploitation equilibrium in 1965 with the corresponding age- and sex-structure at the start of 1987. This start date is arbitrary within the constraint that there is sufficient time from pre-exploitation to the period for which catch and effort data exist (1988-) to tune the model to those catches. The population sizes and the corresponding age- and sex-structures on each reef at the start of the first year are computed using the following algorithm:

1. The number of 20+ cm animals on reef \( r \), \( n_r^+ \), is generated from the lognormal distribution, \( LN(I_1^{r'}, r p^r, 0.5^2) \) where \( r p^r \) is the perimeter of reef \( r \) in km and \( I_1^{r'} \) is a reef-specific tuning parameter. The model assumed the \( I_1^{r'} \) are derived from a relationship between reef density and latitude (Fig. 6.4). The value for \( I_1^{r'} \) is determined by dividing the value from the curve in Fig. 6.4 by the value for a latitude of 16.5 °S (i.e., \( I_1^{r'}=1 \) for reefs at 16.5 °S). This divisor can be modified to achieve different scenarios regarding the status of the resource at the start of 2005. Reef perimeter is used as the measure of habitat because much of the central areas of most reefs in Torres Strait are either emergent consolidated substratum or sand, and so not reef habitat relevant to the target species of the reef line fishery. Reef perimeter is therefore expected to represent the extent of sub-tidal coral reef habitat inhabited by the target species better than (enclosed) reef area. The relationship between perimeter and area of such habitat is likely to vary among reefs but is not well known.
Fig. 6.4. Density of coral trout (20+ cm fish per unit area) from visual surveys and a fitted quadratic curve.

2. The number of 1-year-olds on reef $r$ is then determined using the formula:

$$N_{0,1}^r = n^r / \sum_{a=1}^{x} \tilde{N}_a$$

(6.17)

where $\tilde{N}_a$ is the age-structure of the pre-exploitation population, expressed as a fraction of the number of 1-year-olds, and the summation over age is restricted to fish for which $L_a > 20$ cm. The initial age-structure is computed straightforwardly based on the per-recruit age structure.

3. The biomass corresponding to the generated value for $n^r$ can be such that the population would be extinct prior to the start of the projection period, after all of the historical catch was taken from it. If this occurs, the previous value used for the log-normal mean, (i.e., $I'_r$ $rp'$), is increased by 5% and steps 1 to 3 are repeated since the available catch data are not consistent with such extinctions of coral trout.

6.2.5. Model parameters

The values for the parameters of the biological model are listed in Tables 6.1, 6.2 and 6.3.

Table 6.1. Base-values for the fixed parameters of the biological models for coral trout. Values for the parameters related to changes over time in natural mortality and to catastrophic events are not listed as these factors are not part of the base-case analyses.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maximum age – $x$</td>
<td>18 yr</td>
<td>Mapstone et al. (1996)</td>
</tr>
<tr>
<td>Mortality-at-age (for all ages)</td>
<td>0.45 yr$^{-1}$</td>
<td>Little et al. (2007a, b)</td>
</tr>
<tr>
<td>Temporal variation in natural mortality – $\sigma^M$</td>
<td>0.05</td>
<td>Mapstone et al. (1996)</td>
</tr>
<tr>
<td>Temporal auto-correlation in natural mortality – $\tau^M$</td>
<td>0</td>
<td>Assumed</td>
</tr>
<tr>
<td>Length-mass – $\ell n(h_1)$</td>
<td>-19.2317</td>
<td>Little et al. (2007a, b)</td>
</tr>
<tr>
<td>Length-mass – $h_2$</td>
<td>3.1914</td>
<td>Little et al. (2007a, b)</td>
</tr>
</tbody>
</table>
Larval self seeding – st 0.1 Assumed  
Steepness – h 0.5 Pre-specified  
Maximum age of a ‘juvenile’ - J 18 yr Mapstone et al. (1996)  
Variation in 0-year-old survival – \( \sigma \) 0.6 Mapstone et al. (1996)  
Spatial correlation in 0-year-old survival – \( \tau \) 0.5 Assumed  
Extent of density-dependence in catchability – \( \Phi \) 0 Assumed  
Fraction of fish that die after being released - \( D \) 0.15 Assumed  
Fraction of fish that are retained after being captured - \( R \) 0 Assumed

Table 6.2. Parameters related to selectivity, maturity and the proportion male / sex-change of coral trout.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value (mm FL)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Selectivity</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Length-at-50%-selectivity</td>
<td>322</td>
<td>Fulton et al. (1999)</td>
</tr>
<tr>
<td>Length-at-95%-selectivity</td>
<td>375</td>
<td>Fulton et al. (1999)</td>
</tr>
<tr>
<td>Maturity</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Length-at-50%-maturity</td>
<td>280</td>
<td>S. Adams (pers. comm.)</td>
</tr>
<tr>
<td>Length-at-95%-maturity</td>
<td>360</td>
<td>S. Adams (pers. comm.)</td>
</tr>
<tr>
<td>Sex change</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Length-at-50%-sex change</td>
<td>450</td>
<td>S. Adams (pers. comm.)</td>
</tr>
<tr>
<td>Length-at-95%-sex change</td>
<td>500</td>
<td>S. Adams (pers. comm.)</td>
</tr>
</tbody>
</table>

Table 6.3. von Bertalanffy growth parameters for coral trout.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>( \ell_\infty )</td>
<td>54.05 cm FL</td>
<td>Little et al. (2007a, b)</td>
</tr>
<tr>
<td>( \kappa )</td>
<td>0.339</td>
<td>Little et al. (2007a, b)</td>
</tr>
<tr>
<td>( t_0 )</td>
<td>-0.367</td>
<td>Little et al. (2007a, b)</td>
</tr>
</tbody>
</table>

6.2.6. Effort model

Effort models impose fishing mortality on targeted fish stocks and attempt to represent distributions of fishing effort in response to management strategies (Mapstone et al. 2004). However, modelling the distribution of fishing effort and fisher behaviour has received little attention relative to the investment that has been made in modelling the population dynamics of fish stocks (Hilborn 1985, Holland and Sutinen 1999).

The effort allocation model in ELFSim (Mapstone et al. 2004, Little et al. 2007a,b) is not intended to mimic the decision-making behaviours of individual skippers, but rather to capture the net effect of all such decisions when aggregated to the fleet level. Although the underlying principles of this model are reported in detail elsewhere (see Mapstone et al. 2004), we outline the major facets of the model.

The effort allocation model is based primarily on the historical catch and effort data of the commercial non-indigenous sector. The model allocates fishing effort for this sector over the ETS on a monthly time step during a nominated projection period (i.e., 2005-2025). The total annual effort is divided among the time steps within each year according to intra-annual patterns observed in historical catch and effort data to preserve the seasonality in distribution of fishing effort (see Chapter 3). This is accomplished by selecting a year at random from the period for which there are data (1988-2004), calculating the fraction of that year’s effort that occurred in each month
during the year, and using these fractions to distribute the annual effort among the months during the projection years.

Once the effort available for allocation in each month and sector is determined, it is then allocated among reefs based on historic catch rates, and the management status of each reef (i.e., open or closed to fishing). The management status \( (M) \) of each reef is set between 0 (open) and 1 (closed), with intermediate values capturing the effects of some level of infringement. The effort assigned to a reef is calculated as that effort which would have been allocated to the reef if it was open to fishing multiplied by \( 1-M \). The value of \( M \) can also vary with time, allowing for implementation of temporal closures and temporal variation in infringement.

The effort allocation process then involves ranking reefs that were fished in the previous month according to their weighted CPUE up to that time. The weighted historical CPUE is calculated as the ratio of summed previous catches to summed previous effort over the current calendar month in all previous years (to account for the known seasonality in the dynamics of the fleet), multiplied at each time step by a discount factor (e.g. 0.85) that progressively down-weights older data. The amount of effort allocated to a reef depends on the average historical amount of effort allocated to it. This amount of effort is allocated first to the highest ranked reef and then to the next ranked reefs in turn, until there is no effort left or all of the reefs that were fished in the previous time step have been assigned effort. Any remaining effort is then allocated in the same manner to the reefs that were not fished in the previous time step but had been fished at some earlier time. If unallocated effort still remains after this step, it is allocated randomly to previously unfished reefs in small portions (5% of the balance) until all remaining effort is allocated. This allows for exploration of new fishing grounds and is the only way that effort can be allocated to reefs that have not been fished historically. The catch of coral trout from each reef in each month is then calculated from relevant catch equations.

Effort data for commercial Islander fishers is recorded in freezer logbooks at the coarse spatial scale of Island at which product is sold and not to the scale of individual reefs. Therefore, it was not possible to model the effort of Islander commercial fishers in the same way as the non-indigenous sector. Based on fishing location data, collected at the scale of individual reefs, from access point surveys of Islander commercial fishers (see Chapter 3), commercial Islander fishing effort was modelled statically, and was distributed in proportion to the amount of Islander commercial fishing effort on each reef.

**Management Objectives**

Following the protocols used in the previous coral trout MSE (Mapstone et al. 2004, Little et al. 2007a), we sought input from a range of stakeholders in the ETS RLF to identify relevant management objectives and feasible management strategies by which those objectives may be attained. Specific operational management objectives, performance indicators and alternative management strategies for coral trout were discussed and refined during two Torres Strait Fin Fish Working Group meetings held on Thursday Island in October 2006 and February 2007. Throughout this process our intention was not to seek consensus among the different stakeholders, but to capture their diversity of views (Mapstone et al. 2004), emphasising the benefits of an MSE approach which effectively examines the trade-offs amongst these views.

A total of 45 management objectives were identified for coral trout which varied, across stakeholders, from aspects of conservation to exploitation, and reflected the changing management arrangements within the fishery (Table 6.4). Importantly, the
objectives were specified in terms of quantifiable and measurable indicators that could be evaluated within the ELFSim modelling framework (Table 6.4).

Clarification was also sought on the time periods over which the management objectives were to be assessed. Two time periods were agreed upon: 1) 2011-2015; and 2) 2021-2025. These were considered to represent both short- and mid- to long-term time frames.

Table 6.4. Management objectives and performance indicators for coral trout identified at Torres Strait Fin Fish Working Group meetings. These objectives were evaluated using the ELFSim models and form the basis of the Torres Strait MSE. “Retain” refers to fish caught above MLS (minimum legal size: 38 cm TL). “Total” is all fish kept plus discards. CPUE is catch per unit effort. Total biomass represents fish > 20 cm FL, available biomass represent fish > 38 cm FL and spawning biomass represents all mature female fish. Virgin refers to population levels in 1965, assumed to be unexploited. Time periods evaluated were 2011-2015 and 2021-2025.

<table>
<thead>
<tr>
<th>Objective Number</th>
<th>Management objective</th>
<th>Performance Indicator</th>
<th>Stakeholder</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) ≥ 90% SB&lt;sub&gt;virgin&lt;/sub&gt; 100% of the time</td>
<td>P(SB/SB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.9)</td>
</tr>
<tr>
<td>2</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) ≥ 90% SB&lt;sub&gt;virgin&lt;/sub&gt; 50% of the time</td>
<td>P(SB/SB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.9)</td>
</tr>
<tr>
<td>3</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on closed reefs ≥ 90% SB&lt;sub&gt;virgin&lt;/sub&gt; 100% of the time</td>
<td>P(SB/SB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.9)</td>
</tr>
<tr>
<td>4</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on closed reefs ≥ 90% SB&lt;sub&gt;virgin&lt;/sub&gt; 50% of the time</td>
<td>P(SB/SB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.9)</td>
</tr>
<tr>
<td>5</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on open reefs ≥ 90% SB&lt;sub&gt;virgin&lt;/sub&gt; 100% of the time</td>
<td>P(SB/SB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.9)</td>
</tr>
<tr>
<td>6</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on open reefs ≥ 90% SB&lt;sub&gt;virgin&lt;/sub&gt; 50% of the time</td>
<td>P(SB/SB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.9)</td>
</tr>
<tr>
<td>7</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on open reefs ≥ 40% SB&lt;sub&gt;virgin&lt;/sub&gt; 80% of the time</td>
<td>P(SB/SB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.4)</td>
</tr>
<tr>
<td>8</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) ≥ SB&lt;sub&gt;2004&lt;/sub&gt; 100% of the time</td>
<td>P(SB/SB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
</tr>
<tr>
<td>9</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) ≥ SB&lt;sub&gt;2004&lt;/sub&gt; 50% of the time</td>
<td>P(SB/SB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
</tr>
<tr>
<td>10</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on closed reefs ≥ SB&lt;sub&gt;2004&lt;/sub&gt; 100% of the time</td>
<td>P(SB/SB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
</tr>
<tr>
<td>11</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on closed reefs ≥ SB&lt;sub&gt;2004&lt;/sub&gt; 50% of the time</td>
<td>P(SB/SB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
</tr>
<tr>
<td>12</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on open reefs ≥ SB&lt;sub&gt;2004&lt;/sub&gt; 100% of the time</td>
<td>P(SB/SB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
</tr>
<tr>
<td>13</td>
<td>Spawning Biomass</td>
<td>Spawning biomass (SB) on open reefs ≥ SB&lt;sub&gt;2004&lt;/sub&gt; 50% of the time</td>
<td>P(SB/SB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
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<tr>
<td>Objective Number</td>
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<td>Performance Indicator</td>
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</tr>
<tr>
<td>------------------</td>
<td>----------------------</td>
<td>-----------------------</td>
<td>-------------</td>
</tr>
<tr>
<td>14</td>
<td>Available biomass (AB) ≥ 40% AB&lt;sub&gt;virgin&lt;/sub&gt; 100% of the time</td>
<td>P(AB/AB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.4)</td>
<td>Research</td>
</tr>
<tr>
<td>15</td>
<td>Available biomass (AB) ≥ 40% AB&lt;sub&gt;virgin&lt;/sub&gt; 50% of the time</td>
<td>P(AB/AB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.4)</td>
<td>Research</td>
</tr>
<tr>
<td>16</td>
<td>Available biomass (AB) ≥ 40% AB&lt;sub&gt;virgin&lt;/sub&gt; 80% of the time</td>
<td>P(AB/AB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.4)</td>
<td>AFMA,QDPI&amp;F</td>
</tr>
<tr>
<td>17</td>
<td>Available biomass (AB) on closed reefs ≥ 80% AB&lt;sub&gt;virgin&lt;/sub&gt; 100% of the time</td>
<td>P(AB/AB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.8)</td>
<td>Research</td>
</tr>
<tr>
<td>18</td>
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<td>P(AB/AB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.8)</td>
<td>Research</td>
</tr>
<tr>
<td>19</td>
<td>Available biomass (AB) on open reefs ≥ 80% AB&lt;sub&gt;virgin&lt;/sub&gt; 100% of the time</td>
<td>P(AB/AB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.8)</td>
<td>Research</td>
</tr>
<tr>
<td>20</td>
<td>Available biomass (AB) on open reefs ≥ 80% AB&lt;sub&gt;virgin&lt;/sub&gt; 50% of the time</td>
<td>P(AB/AB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.8)</td>
<td>Research</td>
</tr>
<tr>
<td>21</td>
<td>Available biomass (AB) ≥ AB&lt;sub&gt;2004&lt;/sub&gt; 100% of the time</td>
<td>P(AB/AB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>22</td>
<td>Available biomass (AB) ≥ AB&lt;sub&gt;2004&lt;/sub&gt; 50% of the time</td>
<td>P(AB/AB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>23</td>
<td>Available biomass (AB) on closed reefs ≥ AB&lt;sub&gt;2004&lt;/sub&gt; 100% of the time</td>
<td>P(AB/AB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>24</td>
<td>Available biomass (AB) on closed reefs ≥ AB&lt;sub&gt;2004&lt;/sub&gt; 50% of the time</td>
<td>P(AB/AB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>25</td>
<td>Available biomass (AB) on open reefs ≥ AB&lt;sub&gt;2004&lt;/sub&gt; 100% of the time</td>
<td>P(AB/AB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>26</td>
<td>Available biomass (AB) on open reefs ≥ AB&lt;sub&gt;2004&lt;/sub&gt; 50% of the time</td>
<td>P(AB/AB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>27</td>
<td>Total biomass (TB) ≥ 40% TB&lt;sub&gt;virgin&lt;/sub&gt; 100% of the time</td>
<td>P(TB/TB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.4)</td>
<td>Research</td>
</tr>
<tr>
<td>28</td>
<td>Total biomass (TB) ≥ 40% TB&lt;sub&gt;virgin&lt;/sub&gt; 50% of the time</td>
<td>P(TB/TB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.4)</td>
<td>Research</td>
</tr>
<tr>
<td>29</td>
<td>Total biomass (TB) ≥ 40% TB&lt;sub&gt;virgin&lt;/sub&gt; 80% of the time</td>
<td>P(TB/TB&lt;sub&gt;virgin&lt;/sub&gt; &gt; 0.4)</td>
<td>Non-indigenous Commercial, AFMA, QDPI&amp;F</td>
</tr>
<tr>
<td>30</td>
<td>Total biomass (TB) ≥ TB&lt;sub&gt;2004&lt;/sub&gt; 100% of the time</td>
<td>P(TB/TB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>31</td>
<td>Total biomass (TB) ≥ TB&lt;sub&gt;2004&lt;/sub&gt; 50% of the time</td>
<td>P(TB/TB&lt;sub&gt;2004&lt;/sub&gt; &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>32</td>
<td>Non-indigenous CPUE &gt; 80% 2000-2004 levels 90% of the time</td>
<td>P(CPUE/CPUE&lt;sub&gt;2000-2004&lt;/sub&gt; &gt; 0.8)</td>
<td>Research</td>
</tr>
<tr>
<td>33</td>
<td>Non-indigenous CPUE &gt; 100% 2000-2004 levels 90% of the time</td>
<td>P(CPUE/CPUE&lt;sub&gt;2000-2004&lt;/sub&gt; &gt; 1.0)</td>
<td>AFMA, QDPI&amp;F</td>
</tr>
<tr>
<td>34</td>
<td>Non-indigenous CPUE &gt; 120% 2000-2004 levels 90% of the time</td>
<td>P(CPUE/CPUE&lt;sub&gt;2000-2004&lt;/sub&gt; &gt; 1.2)</td>
<td>Non-indigenous Commercial</td>
</tr>
<tr>
<td>Objective Number</td>
<td>Management objective</td>
<td>Performance Indicator</td>
<td>Stakeholder</td>
</tr>
<tr>
<td>------------------</td>
<td>----------------------</td>
<td>-----------------------</td>
<td>-------------</td>
</tr>
<tr>
<td>35</td>
<td>time</td>
<td>P(CPUE/CPUE\textsubscript{2000-2004} &gt; 0.8)</td>
<td>Islander Commercial Research</td>
</tr>
<tr>
<td>36</td>
<td>time</td>
<td>P(CPUE/CPUE\textsubscript{2000-2004} &gt; 1.2)</td>
<td>Islander Commercial Research</td>
</tr>
<tr>
<td>37</td>
<td>time</td>
<td>P(CPUE/CPUE\textsubscript{2000-2004} &gt; 1.5)</td>
<td>Islander Commercial Research</td>
</tr>
<tr>
<td>38</td>
<td>Harvest</td>
<td>P(\text{retain/total} &gt; 0.5)</td>
<td>Non-indigenous Commercial</td>
</tr>
<tr>
<td>39</td>
<td>Harvest</td>
<td>P(\text{retain/total} &gt; 0.5)</td>
<td>Non-indigenous Commercial</td>
</tr>
<tr>
<td>40</td>
<td>Harvest</td>
<td>P(Harvest/harvest\textsubscript{2000-2004}) &gt; 1</td>
<td>Non-indigenous Commercial</td>
</tr>
<tr>
<td>41</td>
<td>Harvest</td>
<td>P(Harvest/harvest\textsubscript{2000-2004}) &gt; 0.95</td>
<td>Research</td>
</tr>
<tr>
<td>42</td>
<td>Harvest</td>
<td>P(Harvest/harvest\textsubscript{2000-2004}) &gt; 1</td>
<td>Islander Commercial</td>
</tr>
<tr>
<td>43</td>
<td>Harvest</td>
<td>P(Harvest/harvest\textsubscript{2000-2004}) &gt; 0.95</td>
<td>Research</td>
</tr>
<tr>
<td>44</td>
<td>Harvest</td>
<td>P(Harvest/\text{quota} &gt; 1.0)</td>
<td>Research</td>
</tr>
<tr>
<td>45</td>
<td>Sex ratios</td>
<td>P(M/F &gt; M/F\textsubscript{2000-2004})</td>
<td>AFMA</td>
</tr>
</tbody>
</table>

**Management Strategies**

A range of alternative management strategies were also identified and agreed upon by the stakeholders at the Torres Strait Fin Fish Working Group meetings. A total of 16 management strategies were identified, combining spatial and seasonal closures, effort restrictions, and legal sizes (Table 6.5). These strategies were considered by stakeholders to be the most computational and appropriate, considering the management regime under which the ETS RLF operated prior to 2007 (when the meetings were held), while acknowledging the time constraints of evaluating multiple management strategies.
Table 6.5. Management strategies for coral trout identified at Torres Strait Fin Fish Working Group meetings. These strategies were evaluated using the ELFSim models and form the basis of the Torres Strait MSE. Base case refers to current management strategy that the fishery was operating under in 2007.

<table>
<thead>
<tr>
<th>#</th>
<th>Seasonal closure</th>
<th>Spatial closure</th>
<th>Minimum Legal size</th>
<th>Effort/TAC</th>
<th>Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>2 months</td>
<td>10 nm radius</td>
<td>38 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>No closure</td>
<td>10 nm radius</td>
<td>38 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>2 months</td>
<td>No closure</td>
<td>38 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>No closure</td>
<td>No closure</td>
<td>38 cm TL</td>
<td>2004 Effort level</td>
<td>Base case – current (2007) strategies</td>
</tr>
<tr>
<td>5</td>
<td>2 months</td>
<td>10 nm radius</td>
<td>35 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>No closure</td>
<td>10 nm radius</td>
<td>35 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>2 months</td>
<td>No closure</td>
<td>35 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>8</td>
<td>No closure</td>
<td>No closure</td>
<td>35 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>9</td>
<td>2 months</td>
<td>10 nm radius</td>
<td>40 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>No closure</td>
<td>10 nm radius</td>
<td>40 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>11</td>
<td>2 months</td>
<td>No closure</td>
<td>40 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>12</td>
<td>No closure</td>
<td>No closure</td>
<td>40 cm TL</td>
<td>2004 Effort level</td>
<td></td>
</tr>
<tr>
<td>13</td>
<td>No closure</td>
<td>No closure</td>
<td>38 cm TL</td>
<td>80 t (split 50:50 between islander and non-indigenous sector)</td>
<td>Effort required to catch 40 t is allocated to each sector based on historic catch rates</td>
</tr>
<tr>
<td>14</td>
<td>No closure</td>
<td>No closure</td>
<td>38 cm TL</td>
<td>110 t (split 50:50 between islander and non-indigenous sector)</td>
<td>Effort required to catch 55 t is allocated to each sector based on historic catch rates</td>
</tr>
<tr>
<td>15</td>
<td>No closure</td>
<td>No closure</td>
<td>38 cm TL</td>
<td>140 t (split 50:50 between islander and non-indigenous sector)</td>
<td>Effort required to catch 70 t is allocated to each sector based on historic catch rates</td>
</tr>
<tr>
<td>16</td>
<td>No closure</td>
<td>No closure</td>
<td>38 cm TL</td>
<td>170 t (split 50:50 between islander and non-indigenous sector)</td>
<td>Effort required to catch 85 t is allocated to each sector based on historic catch rates</td>
</tr>
</tbody>
</table>

Two levels of seasonal closures were identified: 1) No closures, as is the case under current management arrangements (No closure, in Table 6.5); and 2) A two month block closure to both non-indigenous and Islander sectors between 1st September and 31st October (2 months, in Table 6.5). The two month total closure was considered to best capture the peak spawning periods of coral trout in Torres Strait (Chapter 4). The seasonal closure was implemented simply by removing effort during the closure period, without the redistribution of effort to other months. Thus, the closures act as an effort reduction mechanism.

Two levels of spatial closures were identified: 1) No closures, as is the case under current management arrangements (No closure, in Table 6.5); and 2) a permanent closure to both the non-indigenous and Islander sectors within a 10 nm radius of

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Murray, Darnley, Stephens and Yorke Islands (10 nm radius, in Table 6.5). At the time, we were not able to examine the effects of a rotational closure of these areas to the Islander sector, as it would have added significantly more simulation time. However, we thought the permanent closure to both sectors would give an indication of the maximum effect of such spatial closures.

Three levels of size restrictions on harvest were identified: 1) current minimum legal size of 38 cm TL; 2) a minimum legal size of 35 cm TL; and 3) minimum legal size of 40 cm (Table 6.5). We assumed that fishers would use the same fishing gear (hook size) under each minimum size limit such that selectivity did not differ among size limits.

Two different types of effort restrictions were also identified: 1) 2004 fishing effort (2004 effort level, in Table 6.5). The 2004 fishing effort level was assumed to best represent historical exploitation patterns; and 2) effort required to harvest 4 different levels of Total Allowable Catch (TAC) (80 t, 110 t, 140 t and 170 t, in Table 6.5). The ETS RLF is proposed to move towards a quota managed fishery with a 50:50 quota split between non-indigenous and Islander fishers (e.g. under a TAC of 80 t, each sector would be allocated 40 t). The different levels of TAC were based on a range of potential TACs that are likely for future management of the fishery. Since ELFSim does not currently have the capacity to directly simulate catch quotas, it was necessary to estimate the amount of effort for each sector that would be expected to catch each level of TAC. These effort levels were estimated by dividing each sector component of the TAC by the CPUE for each sector averaged across the years for which data exist (1988-2004). Estimated effort levels for each TAC for each sector are given in Table 6.6.

Table 6.6. Estimated effort levels for the non-indigenous and Islander sectors for a range of Total Allowable Catches (TAC) and reported effort levels for the 2004 catch. TAC levels are for both sectors combined (i.e. 80t TAC = 40t per sector). Estimates are based on sector-specific catch rates derived from the logbook data (Chapter 3) for the non-indigenous (36.30kg/dory day) and Islander (17.45kg/dory day) sectors.

<table>
<thead>
<tr>
<th>Name</th>
<th>Non-indigenous effort (dory days)</th>
<th>Islander effort (dory days)</th>
<th>Equivalent catch</th>
</tr>
</thead>
<tbody>
<tr>
<td>2004 effort level</td>
<td>3007</td>
<td>1545</td>
<td>117t</td>
</tr>
<tr>
<td>E(80)</td>
<td>1102</td>
<td>2292</td>
<td>80t</td>
</tr>
<tr>
<td>E(110)</td>
<td>1515</td>
<td>3152</td>
<td>110t</td>
</tr>
<tr>
<td>E(140)</td>
<td>1928</td>
<td>4011</td>
<td>140t</td>
</tr>
<tr>
<td>E(170)</td>
<td>2342</td>
<td>4871</td>
<td>170t</td>
</tr>
</tbody>
</table>

Model Projections

Management strategies are implemented during the projection period by varying annual effort, access to areas for fishing, and the values of the parameters determining the interaction between the biological and effort models, such as minimum legal sizes. Management strategies can be fully implemented at the start of the projection period or, if they include time-varying measures, during the projections period. However, management strategies are always pre-specified, i.e., there is no dynamic feedback between management strategies and stock dynamics. Evaluations proceed by running the operating model from 1965 to the end of the period for which
real data are supplied (2004) (the initialisation period) and then introducing the desired changes in the parameters that define the management strategy. Random processes in the population dynamics mean that each initialisation will lead to different starting conditions for the projections. The model is then run for a defined projection period (in this case 20 years). Repeating runs with the same management strategy allows an evaluation of the effect of variation in population dynamics and effort allocations on the results for that management strategy. Running the same management strategy with different parameters for the underlying operating model allows an assessment of the robustness of the results to uncertainties or errors in model assumptions. A wide range of reef-specific data can be collected at each time step, including catch and effort for each sector (non-indigenous and islander), available biomass, spawning biomass, fishing mortality and size and age measures for the population and catch. For the purposes of this report we performed 10 initialisations from 1965 to 2004. For each of the 16 management strategies, the 10 initialisations were each followed by 10 replicate projections, resulting in 100 replicate projections for each management strategy.

Analysis of simulation data

The simulation results were analysed and portrayed as follows:

1. The biomass performance indicators (spawning biomass, available biomass and total biomass) were totalled across all reefs (or by open/closed status) for each year. The average of these totals (and standard deviation) were calculated from 2011-2015 and 2021-2025 for each replicate simulation and divided by either the unfished biomass (1965) or 2004 level biomass calculated from the same replicate simulation. The number of these 100 relative biomasses that satisfied the management objective defined the probability that the objective was met. Relative biomass was defined as,

\[
X'_{\text{sim}} = \frac{1}{n_y \sum_r \sum \tau y \tau r X_{\text{sim},y,r}} X_{\text{sim}, \text{ref}, r}
\]

where \(X'_{\text{sim}}\) is the relative biomass (spawning, available or total) in replicate simulation \(\text{sim}\),

\(X_{\text{sim}, y, r}\) is the biomass (spawning, available or total) in replicate simulation \(\text{sim}\) in year \(y\) (\(\text{ref}\) is reference year) on reef \(r\).

2. The CPUE indicators were similarly calculated by totalling catch and effort across all reefs for all years, and then dividing the total catch by the total effort to obtain an annual CPUE. The average and standard deviation of the annual CPUE was calculated across the years from 2011-2015 and 2021-2025 for the projection period, and across the historical reference years 1988-2004 for each sector. Relative CPUE was calculated by dividing the average historical CPUE into the average projected CPUE from 2011-2015 or 2021-2025. Of the 100 relative CPUEs the number that satisfied the management objective defined the probability that the objective was met:
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\[
CPUE_{sim}^r = \frac{1}{n_y} \sum_{y_1}^{y_2} \frac{\sum_{r}^{r_1} C_{sim,y_1,r}}{\sum_{r}^{r_1} E_{sim,y_1,r}} = \frac{1}{n_y} \sum_{y_1}^{y_2} \frac{\sum_{r}^{r_1} C_{sim,y_1,r}}{\sum_{r}^{r_1} E_{sim,y_1,r}}
\]

where \( CPUE_{sim}^r \) is the relative CPUE in replicate simulation \( sim \)

\( C_{sim,y,r} \) is the catch in replicate simulation \( sim \), in year \( y \) (\( ref \) is reference year) on reef \( r \)

\( E_{sim,y,r} \) is the effort in replicate simulation \( sim \), in year \( y \) (\( ref \) is reference year) on reef \( r \)

\( y_1 \) is the summation interval for the projection period

\( y_2 \) is the summation interval for the historical period

3. The harvest indicators were calculated by adding the amount of fish released and the amount of fish retained across all reefs. These values were averaged across the years 2011-2015 and 2021-2025. The performance indicator that concerned legal retentions was calculated by dividing the average amount of fish retained across all reefs during the respective interval (2011-2015 or 2021-2025) by the average amount of fish caught (released plus retained) during the respective interval. Similarly, the performance indicator that concerned quota was calculated by dividing the average amount of fish retained across all reefs during the respective interval by the relevant quota level. The proportion of the 100 replicated simulations that satisfied the management objective defined the probability that the objective was met.

\[
C_{sim}^r = \frac{1}{n_y} \sum_{y_1}^{y_2} \sum_{r}^{r_1} C_{sim,y_1,r} \frac{1}{K^*}
\]

where \( C_{sim}^r \) is the relative harvest indicator in replicate simulation \( sim \)

\( C_{sim,y,r} \) is the retained catch in replicate simulation \( sim \), in year \( y \) (\( ref \) is reference year) on reef \( r \)

\( K^* \) is either the quota level or the average total amount of fish caught

\[
\frac{1}{n_y} \sum_{y_1}^{y_2} \sum_{r}^{r_1} C_{sim,y_1,r} + C_{sim,y_1,r}^R
\]
\[ C_{\text{sim}, y, r}^R \text{ is the released catch in replicate simulation } \text{sim}, \text{ in year } y \text{ (ref is reference year) on reef } r. \]

4. The size indicator was calculated by averaging the proportion of fish (either <1.2kg or < 1.5kg) in the catch across all reefs for each year of the projection period. These were then averaged across the years 2011-2015 and 2021-2025:

\[ B_{\text{sim}}^* = \frac{1}{n_{y_1}} \sum_{y} \sum_{r} B_{\text{sim}, y, r}^* \]  

(6.21)

where \( B_{\text{sim}}^* \) is the size indicator for fish either <1.2 kg or < 1.5kg in replicate simulation \( \text{sim} \)

\( B_{\text{sim}, y, r}^* \) is the proportion of fish either <1.2 kg or < 1.5kg on reef \( r \), in year \( y \) and replicate simulation \( \text{sim} \)
6.3. Results

6.3.1. Spawning Biomass
Spawning biomass increased from 2004 levels under all management strategy combinations (Fig. 6.5), including the current management strategy of no closures and 38 cm MLS (Fig. 6.5E). The increase in spawning biomass was generally slightly greater with some form of closures (Fig. 6.5A-C) than with no closures at all (Fig. 6.5E). Spawning biomass responded similarly to spatial and seasonal closures, although it responded more quickly to seasonal closures (Fig. 6.5A,B). The combination of spatial and seasonal closures did not result in greater spawning biomass than when spatial or seasonal closures were examined separately (Fig. 6.5C). Increasing the MLS generally resulted in greater spawning biomass under all spatial and seasonal closure scenarios. As expected, increasing effort generally resulted in lower spawning biomass, although all effort levels resulted in stable or increasing trend in spawning biomass (Fig. 6.5D).

Relative spawning biomass at the middle and end of the projection period (2011-2015 and 2021-2025) was higher with seasonal or spatial closures than without closures, although the effect was only slight (Fig. 6.6). Changes in the MLS and effort level had a slightly greater effect on spawning biomass than seasonal or spatial closures (Fig. 6.6). Spawning biomass increased with an increase in the MLS and declined with increasing effort levels.

When projected spawning biomass was considered relative to that projected under the 'status quo' management scenario (no spatial or seasonal closures, 38 cm MLS and 2004 effort level of 117 t), greater differences among management strategies were evident (Fig. 6.7). Increasing the MLS from 38 cm to 40 cm increased the amount of spawning biomass, while reducing the MLS to 35 cm resulted in a lower spawning biomass than under status quo management (Fig. 6.7). Seasonal and spatial closures, in combination with a 38 cm or 40 cm MLS, resulted in greater spawning biomass than status quo management, although seasonal closures had a slightly greater effect than spatial closures (Fig. 6.7). All effort scenarios resulted in greater spawning biomass than status quo management strategies, except the E(170 t) effort level which resulted in a lower spawning biomass (Fig. 6.7).

The first objective concerning spawning biomass (Objective 1, Table 6.4) was that it should be above 90% of the unfished, pre-exploitation, levels 100% of the time. A second objective (objective 2, Table 6.4) was that it should be above 90% of the unfished, pre-exploitation, levels 50% of the time. Figure 6.8 shows that no management strategy achieved either of these objectives. A seasonal closure and 40 cm minimum legal size and the E(80 t) effort level were the management strategies that came closest to meeting the second objective.
A) Spatial closure

B) Seasonal closure

C) Spatial and seasonal closures

D) Total Allowable Catch

E) No closures

Fig. 6.5. Trajectories of spawning biomass relative to virgin spawning biomass averaged over 10 simulations for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
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Fig. 6.6. Average (±SE) spawning biomass (2011-2015 black bars and 2021-2025 grey bars) as a proportion of virgin (unfished) spawning biomass.

Fig. 6.7. Average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) relative to spawning biomass under the current management strategy.
Management Strategy Evaluation

Fig. 6.8. Proportion of simulations where the average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) exceeds 90% of the unfished spawning biomass. Objective 1 (100% of the time) dashed line, objective 2 (50% of the time) solid line.

Stakeholders were also interested in the specific effects of different management strategies on reefs open and closed to fishing. When examining the results of the simulations on closed reefs, only management strategy combinations that involved spatial closures are relevant, since they are the only ones that contain reefs where fishing was precluded. Changes in MLS were the only management strategy combined with spatial closures, as the different effort levels were simulated with no spatial closures. Projected spawning biomass on closed reefs was only marginally higher with seasonal closures than without closures (Fig. 6.9). Increasing the MLS also resulted in slightly higher spawning biomass on closed reefs at the end of the project periods (Fig. 6.9).

Spawning biomass on closed reefs was equal to or greater than that under status quo management for all management strategies (Fig. 6.10). Seasonal closures and an increase in the MLS both resulted in greater spawning biomass on closed reefs relative to status quo management, although a change in MLS had a greater effect (Fig. 6.10).

No management strategies were successful in achieving the objective of maintaining spawning biomass on closed reefs above 90% of virgin levels 100% of the time (Objective 3, Fig. 6.11). An increase in the MLS to 40 cm was the only management strategy that achieved this objective more than 50% of the time (Objective 4, Fig. 6.11). However, the current MLS of 38 cm came close to achieving this objective, particularly at the end of the projection period (2021-2025).
Fig. 6.9. Average (±SE) spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on the closed reefs as a proportion of virgin (unfished) spawning biomass on the same reefs.

Fig. 6.10. Average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on closed reefs relative to spawning biomass under the current management strategy.
Fig. 6.11. Proportion of simulations where the average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on the closed reefs exceeds 90% of the unfished spawning biomass on the same reefs. Objective 3 (100% of the time) dashed line, objective 4 (50% of the time) solid line.

The effects of different management strategies on spawning biomass were similar for reefs open to fishing as for reefs closed to fishing. Projected spawning biomass on open reefs was marginally higher with seasonal closures than without closures (Fig. 6.12). However, spawning biomass on open reefs was similar with and without spatial closures (Fig. 6.12). Increasing the MLS and decreasing the effort level both resulted in higher spawning biomass on open reefs at the end of the projection periods (Fig. 6.12).

Greater differences among management strategies were evident when spawning biomass on open reefs was considered relative to status quo management (Fig. 6.13). Spawning biomass on open reefs was always lower with spatial closures than without, and in nearly all cases, spawning biomass on open reefs was lower with spatial closures than under status quo management (Fig. 6.13). Seasonal closures and an increase in the MLS both resulted in greater spawning biomass on open reefs relative to status quo management, although a change in MLS had a greater effect (Fig. 6.13). All effort scenarios resulted in greater spawning biomass on open reefs than status quo management strategies, except the E(170 t) effort level which resulted in a lower spawning biomass (Fig. 6.13).

No management strategies were successful in achieving the objective of maintaining spawning biomass on open reefs above 90% of virgin levels 100% (Objective 5) or 50% (Objective 6) of the time (Fig. 6.14). An increase in the MLS to 40 cm and an effort level of E(80 t) were the management strategies that came closest to achieving these objectives (Fig. 6.14).

An additional objective (Objective 7) was that spawning biomass on open reefs should be above 40% of the unfished levels 80% of the time. All management
strategies achieved this objective and, in fact, all achieved it 100% of the time (Fig. 6.15).

![Graph showing mean spawning biomass on open reefs as a proportion of unfished spawning biomass on the same reefs.]

**Fig. 6.12.** Average (±SE) spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs as a proportion of virgin (unfished) levels of spawning biomass on the same reefs.

![Graph showing mean spawning biomass on open reefs relative to current management strategy.]

**Fig. 6.13.** Average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs relative to spawning biomass under the current management strategy.
Fig. 6.14. Proportion of simulations where the average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs exceeds 90% of the unfished spawning biomass on the same reefs. Objective 5 (100% of the time) dashed line, Objective 6 (50% of the time) solid line.

Fig. 6.15. Proportion of simulations where the average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs exceeds 40% of the unfished spawning biomass on the same reefs. Objective 7 (80% of the time) solid line.
The management objectives concerning spawning biomass related not only to the pre-exploitation or unfished levels, but also to the spawning biomass levels in 2004 (Table 6.4). Average spawning biomass across all reefs (i.e., reefs open and closed to fishing) ranged from 1.16 to 1.45 of the 2004 spawning biomass (Fig. 6.16). Spawning biomass increased slightly with seasonal closures, but there was no consistent trend in spawning biomass with spatial closures (Fig. 6.16). Increasing the MLS and decreasing the effort level both resulted in higher spawning biomass at the middle and end of the projection period (Fig. 6.16).

All of the strategies led to spawning biomass greater than the 2004 level more than 50% (Objective 9) of the time, and almost all strategies met this objective 100% (Objective 8) of the time (Fig. 6.17). The only exception was the combined effect of a 35 cm MLS, spatial closures and no seasonal closures.

Seasonal closures and MLS were the only management strategies involving reefs closed to fishing. Average spawning biomass on reefs closed to fishing ranged from 1.34 to 1.50 of the 2004 spawning biomass on these reefs (Fig. 6.18). Spawning biomass increased only slightly with seasonal closures and with increasing MLS (Fig. 6.18).

All of the strategies led to spawning biomass on closed reefs greater than the 2004 level on these reefs for more than 50% (Objective 11) and 100% (Objective 10) of the time (Fig. 6.19).

Fig. 6.16. Average (±SE) spawning biomass (2011-2015 black bars and 2021-2025 grey bars) as a proportion of the 2004 spawning biomass.
Fig. 6.17. Proportion of simulations where the average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) exceeds the 2004 spawning biomass. Objective 8 (100% of the time) dashed lined, Objective 9 (50% of the time) solid line.

Fig. 6.18. Average (±SE) spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on closed reefs as a proportion of the 2004 spawning biomass on the same reefs.
Fig. 6.19. Proportion of simulations where the average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on closed reefs exceeds the 2004 spawning biomass on the same reefs. Objective 10 (100% of the time) dashed line, Objective 11 (50% of the time) solid line.

Average spawning biomass on reefs open to fishing ranged from 1.15 to 1.45 of the 2004 spawning biomass on these reefs (Fig. 6.20). Spawning biomass on open reefs increased slightly with seasonal closures, but there was no consistent trend in spawning biomass with spatial closures (Fig. 6.20). Increasing the MLS and decreasing the effort level both resulted in higher spawning biomass on open reefs at the middle and end of the projection period (Fig. 6.20).

Similar to the simulations for all reefs, all of the strategies led to spawning biomass on open reefs greater than the 2004 level more than 50% (Objective 13) of the time, and almost all strategies met this objective 100% (Objective 12) of the time (Fig. 6.21). The only exception was the combined effect of a 35 cm MLS, spatial closures and no seasonal closures.
Fig. 6.20. Average (±SE) spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs as a proportion of the 2004 spawning biomass on the same reefs.

Fig. 6.21. Proportion of simulations where the average spawning biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs exceeds the 2004 spawning biomass on the same reefs. Objective 12 (100% of the time) dashed lined, Objective 13 (50% of the time) solid line.
6.3.2. Available Biomass

Similar to spawning biomass, available biomass increased from 2004 levels under all management strategy combinations (Fig. 6.22), including the current management strategy of no closures and 38 cm MLS (Fig. 6.22E). Available biomass responded more quickly to seasonal closures and reached a higher level than for spatial closures (Fig. 6.22 A,B). Interestingly, available biomass was generally lower with spatial closures (Fig. 6.22A) than with no spatial or temporal closures (Fig. 6.22E). The combination of spatial and seasonal closures did not result in greater available biomass than when spatial or seasonal closures were examined separately (Fig. 6.22C). In contrast to spawning biomass, increasing the MLS from 38 cm to 40 cm did not result in greater available biomass. A MLS of 38 cm resulted in the highest available biomass, while a MLS of 40 cm resulted in the lowest available biomass under all spatial and seasonal closure scenarios. There was an initial sharp decline in available biomass for the first projection year, before a gradual increase, for the 40 cm MLS management strategy in combination with all spatial and seasonal closures (Fig. 6.22A-C). This decline is due to the fact that available biomass prior to 2004 was defined in the model as biomass of fish greater than 38 cm (not 40 cm), such that projected available biomass for the 35 cm and 40 cm scenarios will be relative to available biomass of fish greater than 38 cm (not 40 cm or 35 cm). As expected, increasing effort generally resulted in lower available biomass, although all effort levels resulted in stable or increasing trend in available biomass (Fig. 6.22D).

Relative available biomass at the middle and end of the projection period (2011-2015 and 2021-2025) was higher with seasonal or spatial closures than without closures, although the effect was only slight (Fig. 6.23). Changes in the MLS and effort level had a greater effect on available biomass than seasonal or spatial closures (Fig. 6.23). Available biomass declined with increasing effort levels and was greatest for the current MLS of 38 cm and lowest for a MLS of 40 cm.

When projected available biomass was considered relative to that projected under the status quo management scenario, greater differences among management strategies were evident (Fig. 6.24). In contrast to spawning biomass, increasing the MLS from 38 cm to 40 cm significantly reduced the amount of available biomass, while reducing the MLS to 35 cm also resulted in lower available biomass than under status quo management (Fig. 6.24). Seasonal and spatial closures, in combination with the current 38 cm MLS, resulted in greater available biomass than status quo management, although seasonal closures had a slightly greater effect than spatial closures (Fig. 6.24). Similar to spawning biomass, all effort scenarios resulted in greater available biomass than status quo management strategies, except the E(170 t) effort level which resulted in a lower available biomass (Fig. 6.24).

The first three objectives concerning available biomass were that it should be above 40% of the unfished, pre-exploitation, levels 100% (Objective 14), 50% (Objective 15) or 80% (Objective 16) of the time. Figure 6.25 shows that all management strategies achieved these three objectives, except those management strategies concerning a MLS of 40 cm. Seasonal and spatial closures ameliorated the effects of the 40 cm MLS such that available biomass was greater than 40% unfished levels at least 50% of the time (Figure 6.25).
Management Strategy Evaluation

A) Spatial closure

B) Seasonal closure

C) Spatial and seasonal closures

D) Total Allowable Catch

E) No closures

Fig. 6.22. Trajectories of available biomass relative to virgin available biomass averaged over 10 simulations for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
Fig. 6.23. Average (±SE) available biomass (2011-2015 black bars and 2021-2025 grey bars) as a proportion of virgin (unfished) available biomass.

Fig. 6.24. Average available biomass (2011-2015 black bars and 2021-2025 grey bars) relative to available biomass under the current management strategy.
Projected available biomass on closed reefs was only marginally higher with seasonal closures than without closures (Fig. 6.26). MLS had a greater effect on available biomass, which was greatest for the 38 cm MLS and lowest for the 40 cm MLS (Fig. 6.26).

Available biomass on closed reefs was equal to or greater than that under status quo management for the 35 cm and 38 cm MLS, but substantially lower for the 40 cm MLS (Fig. 6.27). Seasonal closures resulted in only a slightly greater available biomass on closed reefs relative to status quo management (Fig. 6.27).

No management strategies were successful in achieving the objective of maintaining available biomass on closed reefs above 80% of virgin levels 100% of the time (Objective 17, Fig. 6.28). A MLS of 35 cm or 38 cm both achieved this objective more than 50% of the time (Objective 18, Fig. 6.28). An increase in the MLS to 40 cm failed to achieve either objective.
Fig. 6.26. Average (±SE) available biomass (2011-2015 black bars and 2021-2025 grey bars) on the closed reefs as a proportion of virgin (unfished) available biomass on the same reefs.

Fig. 6.27. Average available biomass (2011-2015 black bars and 2021-2025 grey bars) on closed reefs relative to available biomass under the current management strategy.
Projected available biomass on open reefs was higher with seasonal closures than without closures (Fig. 6.29). However, available biomass on open reefs was marginally higher with spatial closures than without (Fig. 6.29). A MLS of 38 cm resulted in the highest available biomass, while a MLS of 40 cm resulted in the lowest available biomass (Fig. 6.29). Decreasing the effort level resulted in higher available biomass on open reefs at the middle and end of the projection period (Fig. 6.29).

Greater differences among management strategies were evident when available biomass on open reefs was considered relative to status quo management (Fig. 6.30). Available biomass on open reefs exceeded levels under status quo management only for management strategies with seasonal closures, no spatial closures and a MLS of 35 or 38 cm and for all effort levels except E(170 t) (Fig. 6.30).

No management strategies were successful in achieving the objective of maintaining available biomass on open reefs above 80% of virgin levels 100% (Objective 19) or 50% (Objective 20) of the time (Fig. 6.31). An effort level of E(80 t) was the only management strategy that achieved these objectives, but only 3% of the time (Fig. 6.31).
Fig. 6.29. Average (±SE) available biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs as a proportion of virgin (unfished) levels of available biomass on the same reefs.

Fig. 6.30. Average available biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs relative to available biomass under the current management strategy.
Similar to spawning biomass, the management objectives concerning available biomass related not only to the pre-exploitation or unfished levels, but also to the available biomass levels in 2004 (Table 6.4). Average available biomass across all reefs (i.e., reefs open and closed to fishing) ranged from 1.05 to 1.84 of the 2004 available biomass (Fig. 6.32). Available biomass increased with seasonal closures, but there was no consistent trend in available biomass with spatial closures (Fig. 6.32). A MLS of 38 cm resulted in the highest available biomass, while a MLS of 40 cm resulted in the lowest available biomass relative to 2004 levels. Decreasing the effort level resulted in higher available biomass at the middle and end of the projection period relative to 2004 levels (Fig. 6.32).

All of the management strategies led to available biomass greater than the 2004 level more than 50% (Objective 22) of the time. Almost all strategies met this objective 100% (Objective 21) of the time, except those management strategies that involved a MLS of 40 cm (Fig. 6.33).
**Evaluation of the eastern Torres Strait reef line fishery**

**Fig. 6.32.** Average (±SE) available biomass (2011-2015 black bars and 2021-2025 grey bars) as a proportion of the 2004 available biomass.

**Fig. 6.33.** Proportion of simulations where the average available biomass (2011-2015 black bars and 2021-2025 grey bars) exceeds the 2004 available biomass. Objective 21 (100% of the time) dashed lined, Objective 22 (50% of the time) solid line.
Seasonal closures and MLS were the only management strategies involving reefs closed to fishing. Average available biomass on reefs closed to fishing ranged from 1.46 to 2.03 of the 2004 available biomass on these reefs (Fig. 6.34). Available biomass increased only slightly with seasonal closures (Fig. 6.34). A MLS of 38 cm resulted in the greatest available biomass, while a MLS of 40 cm resulted in the lowest available biomass relative to 2004 levels (Fig. 6.34).

All of the strategies led to available biomass on closed reefs greater than the 2004 level on these reefs for more than 50% (Objective 24) and 100% (Objective 23) of the time (Fig. 6.35).

**Fig. 6.34.** Average (±SE) available biomass (2011-2015 black bars and 2021-2025 grey bars) on closed reefs as a proportion of the 2004 available biomass on the same reefs.
Average available biomass on reefs open to fishing ranged from 0.95 to 1.84 of the 2004 available biomass on these reefs (Fig. 6.36). Available biomass on open reefs increased slightly with seasonal closures, but there was no consistent trend in available biomass with spatial closures (Fig. 6.36). Available biomass on open reefs was greatest for a MLS of 38 cm and lowest for 40 cm relative to 2004 levels (Fig. 6.36). Decreasing the effort level resulted in higher available biomass on open reefs at the middle and end of the projection period relative to 2004 levels (Fig. 6.36).

Similar to the simulations for all reefs, all of the management strategies led to available biomass on open reefs greater than the 2004 level more than 50% (Objective 26) of the time, and almost all strategies met this objective 100% (Objective 25) of the time (Fig. 6.37). The only exception was those management strategies that involved a MLS of 40 cm (Fig. 6.37).

**Fig. 6.35.** Proportion of simulations where the average available biomass (2011-2015 black bars and 2021-2025 grey bars) on closed reefs exceeds the 2004 available biomass on the same reefs. Objective 23 (100% of the time) dashed lined, Objective 24 (50% of the time) solid line.
Fig. 6.36. Average (±SE) available biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs as a proportion of the 2004 available biomass on the same reefs.

Fig. 6.37. Proportion of simulations where the average available biomass (2011-2015 black bars and 2021-2025 grey bars) on open reefs exceeds the 2004 available biomass on the same reefs. Objective 25 (100% of the time) dashed lined, Objective 26 (50% of the time) solid line.
6.3.3. Total Biomass

Similar to spawning and available biomass, total biomass increased from 2004 levels under all management strategy combinations (Fig. 6.38), including the current management strategy of no closures and 38 cm MLS (Fig. 6.38E). Total biomass responded more quickly to seasonal closures and reached a higher level than for spatial closures (Fig. 6.38A,B). Trajectories of available biomass were similar when there were spatial closures (Fig. 6.38A) as when there were no spatial or temporal closures (Fig. 6.38E). The combination of spatial and seasonal closures did not result in greater total biomass than when spatial or seasonal closures were examined separately (Fig. 6.38C). Similar to spawning biomass, increasing the MLS generally resulted in greater total biomass under all spatial and seasonal closure scenarios (Fig. 6.38A-C). As expected, increasing effort generally resulted in lower total biomass, although all effort levels resulted in stable or increasing trend in total biomass (Fig. 6.38D).

Relative total biomass at the middle and end of the projection period (2011-2015 and 2021-2025) was higher with seasonal or spatial closures than without closures, although the effect was only slight (Fig. 6.39). Changes in the MLS and effort level had a greater effect on total biomass than seasonal or spatial closures (Fig. 6.39). Total biomass increased with an increase in the MLS and declined with increasing effort levels.

When projected total biomass was considered relative to that projected under the status quo management scenario, greater differences among management strategies were evident (Fig. 6.40). Most management strategies resulted in greater total biomass than under status quo management (Fig. 6.40). The only exceptions were for the E(170 t) effort level and a MLS of 35 cm in combination with no seasonal closures (Fig. 6.40). A MLS of 40 cm and low effort levels resulted in the greatest total biomass relative to status quo management (Fig. 6.40).

The first three objectives concerning total biomass were that it should be above 40% of the unfished, pre-exploitation, levels 100% (Objective 27), 50% (Objective 28) or 80% (Objective 29) of the time. Figure 6.41 shows that all management strategies achieved these three objectives.
Fig. 6.38. Trajectories of total biomass relative to virgin total biomass averaged over 10 simulations for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
Fig. 6.39. Average (±SE) total biomass (2011-2015 black bars and 2021-2025 grey bars) as a proportion of virgin (unfished) total biomass.

Fig. 6.40. Average total biomass (2011-2015 black bars and 2021-2025 grey bars) relative to total biomass under the current management strategy.
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Fig. 6.41. Proportion of simulations where the average total biomass (2011-2015 black bars and 2021-2025 grey bars) exceeds 40% of unfished total biomass. Objective 27 (100% of the time) dashed lined, Objective 28 (50% of the time) solid line and Objective 29 (80% of the time) dotted line.

Similar to spawning and available biomass, the management objectives concerning total biomass related not only to the pre-exploitation or unfished levels, but also to the total biomass levels in 2004 (Table 6.4). Average total biomass across all reefs ranged from 1.24 to 1.66 of the 2004 total biomass (Fig. 6.42). Total biomass generally increased with both seasonal and spatial closures except under a MLS of 38 cm when spatial closures resulted in a slightly lower total biomass at the end of the projection period (2021-2025) (Fig. 6.42). Increasing the MLS and decreasing the effort level both resulted in higher total biomass at the middle and end of the projection period relative to 2004 levels (Fig. 6.42).

All of the management strategies led to total biomass greater than the 2004 level more than 50% (Objective 31) and 100% (Objective 30) of the time (Fig. 6.43).
**Fig. 6.42.** Average (±SE) total biomass (2011-2015 black bars and 2021-2025 grey bars) as a proportion of the 2004 total biomass.

**Fig. 6.43.** Proportion of simulations where the average total biomass (2011-2015 black bars and 2021-2025 grey bars) exceeds the 2004 total biomass. Objective 30 (100% of the time) dashed lined, Objective 31 (50% of the time) solid line.
6.3.4. Catch per Unit Effort

Trajectories of CPUE for non-indigenous commercial fishers dropped in the first year of projections (2005) to around 37-51% of the average 2000-2004 levels under all management strategies (Fig. 6.44). Trajectories of CPUE under all minimum legal size (MLS) scenarios indicated that CPUE would gradually increase from levels in 2005 to between 50% and 70% of what it had been on average between 2000 and 2004 (Fig. 6.44A-C, E). Note that CPUE in 2000 and 2001 were substantially higher than in any other year. Seasonal closures (Fig. 6.44B) resulted in slightly higher CPUE than spatial closures (Fig. 6.44A) and no closures at all (Fig. 6.44E). The combination of seasonal and spatial closures did not result in higher CPUE than when spatial or seasonal closures were examined separately (Fig. 6.44C). Increasing the MLS generally resulted in lower CPUE under all spatial and seasonal closure scenarios (Fig. 6.44A-C, E). As expected, increasing effort resulted in lower CPUE, although all effort levels resulted in a stable CPUE at or above 2004 levels (Fig. 6.44D).

CPUE for non-indigenous fishers at the middle and end of the projection period (2011-2015 and 2021-2025) was higher with seasonal closures than without closures (Fig. 6.45). However, CPUE was lower with spatial closures than without closures. Increasing the MLS or effort level both resulted in lower CPUE, although the effect of effort level was much greater than MLS (Fig. 6.45).

Most management strategies resulted in higher CPUE for non-indigenous commercial fishers than under status quo management (Fig. 6.46). The only exceptions were for no seasonal closures in combination with a MLS of 38 cm or 40 cm (Fig. 6.46). Low effort levels resulted in the highest CPUE relative to status quo management (Fig. 6.46).

The first three objectives concerning CPUE for non-indigenous fishers were that it should be above 80% (Objective 32), 100% (Objective 33) or 120% (Objective 34) of 2000-2004 levels 90% of the time. No management strategies were able to achieve any of these three objectives (Figs. 6.47 and 6.48) and no management strategy achieved 120% 2000-2004 CPUE levels in any simulation. An E(80 t) effort level was the management strategy that came closest to achieving any of these objectives.
Fig. 6.44. Trajectories of mean catch per unit of effort (CPUE) by non-indigenous commercial fishers relative to mean CPUE between 2000 and 2004 averaged over 10 simulations for reefs open to fishing for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
Fig. 6.45. Average (±SE) catch per unit of effort (CPUE) by non-indigenous commercial fishers (2011-2015 black bars and 2021-2025 grey bars) as a proportion of the average CPUE between 2000 and 2004.

Fig. 6.46. Average (±SE) catch per unit of effort (CPUE) by non-indigenous commercial fishers (2011-2015 black bars and 2021-2025 grey bars) as a proportion of the average CPUE between 2000 and 2004 under the current management strategy.
Fig. 6.47. Proportion of simulations where the catch per unit of effort (CPUE) by non-indigenous commercial fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 80% of the average CPUE between 2000 and 2004 (Objective 32) 90% of the time.

Fig. 6.48. Proportion of simulations where the catch per unit of effort (CPUE) by non-indigenous commercial fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 100% of the average CPUE between 2000 and 2004 (Objective 33) 90% of the time.
Similar to non-indigenous CPUE, trajectories of CPUE for Islander commercial fishers dropped substantially in the first year of projections (2005) to around 34-54% of the average 2000-2004 levels under all management strategies (Fig. 6.49). In the following projection years, CPUE initially increased rapidly then gradually from levels in 2005 under all management strategies (Fig. 6.49). Trajectories of catch per unit of fishing effort (CPUE) for Islander commercial fishers were similar when there were spatial closures (Fig. 6.49A) as when there were no spatial or temporal closures (Fig. 6.49E). CPUE reached higher levels and responded faster to seasonal closures (Fig. 6.49B) than spatial closures (Fig. 6.49A), although the combination of seasonal and spatial closures resulted in lower CPUE than when spatial or seasonal closures were examined separately (Fig. 6.49C). Increasing the MLS generally resulted in lower CPUE under all spatial and seasonal closure scenarios (Fig. 6.49A-C, E). As expected, increasing effort resulted in lower CPUE, although all effort levels resulted in a stable CPUE (Fig. 6.49D).

Similar to non-indigenous fishers, the CPUE for Islander fishers at the middle and end of the projection period (2011-2015 and 2021-2025) was higher with seasonal closures than without closures (Fig. 6.50). However, CPUE was significantly lower with spatial closures than without closures. Increasing the MLS or effort level both resulted in lower CPUE, although the effect of effort level was much greater than MLS (Fig. 6.50).

In contrast to non-indigenous fishers, most management strategies resulted in lower CPUE for Islander commercial fishers than under status quo management (Fig. 6.51). Seasonal closures and no spatial closures in combination with a MLS of 38 cm or 40 cm resulted in the highest CPUE relative to status quo management (Fig. 6.51). The higher effort levels resulted in the lowest CPUE relative to status quo management (Fig. 6.51).

The first three objectives concerning CPUE for Islander fishers were that it should be above 80% (Objective 35), 120% (Objective 36) or 150% (Objective 37) of 2000-2004 levels 90% of the time. No management strategies were able to achieve any of these three objectives (Fig. 6.52) and no management strategy achieved 120% or 150% 2000-2004 CPUE levels in any simulation. Seasonal closures and no spatial closures in combination with a MLS of 38 cm or 40 cm were the management strategies that came closest to reaching 80% 2000-2004 CPUE 90% of the time (Fig. 6.52).
Fig. 6.49. Trajectories of mean catch per unit of effort (CPUE) by islander commercial fishers relative to mean CPUE between 2000 and 2004 averaged over 10 simulations for reefs open to fishing for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
**Fig. 6.50.** Average (±SE) catch per unit of effort (CPUE) by Islander commercial fishers (2011-2015 black bars and 2021-2025 grey bars) as a proportion of the average CPUE between 2000 and 2004.

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<th>2004 Effort level</th>
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**Fig. 6.51.** Average (±SE) catch per unit of effort (CPUE) by Islander commercial fishers (2011-2015 black bars and 2021-2025 grey bars) as a proportion of the average CPUE between 2000 and 2004 under the current management strategy.

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<th>Management Strategy</th>
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<th>Spatial closure</th>
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<th>Effort</th>
<th>2004 Effort level</th>
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Fig. 6.52. Proportion of simulations where the catch per unit of effort (CPUE) by Islander commercial fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 80% of the average CPUE between 2000 and 2004 (Objective 32) 90% of the time.

6.3.5. Harvest

Similar to CPUE, trajectories of harvest for non-indigenous commercial fishers dropped substantially in the first year of projections (2005) to around 29-78% of the average 2000-2004 levels under all management strategies (Fig. 6.53). Trajectories of harvest indicated that, under all MLS scenarios, harvest gradually increased from levels in 2005 to between 75% and 95% of what it had been on average between 2000 and 2004 (Fig. 6.53A-C, E). Spatial closures (Fig. 6.53A) and no closures (Fig. 6.53E) resulted in slightly higher harvest than seasonal closures (Fig. 6.53B). The combination of seasonal and spatial closures resulted in lower harvest than when spatial or seasonal closures were examined separately (Fig. 6.53C). Increasing the MLS generally resulted in lower harvest under all spatial and seasonal closure scenarios (Fig. 6.53A-C, E). As expected, increasing effort resulted in greater harvest relative to 2000-2004 levels (Fig. 6.53D).
Fig. 6.53. Trajectories of mean harvest by non-indigenous commercial fishers relative to mean harvest between 2000 and 2004 averaged over 10 simulations for reefs open to fishing for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
Trajectories of harvest for Islander commercial fishers dropped in the first year of projections (2005) to around 52-97% of the average 2000-2004 levels under all combinations of spatial and seasonal closures (Fig. 6.53A-C, E). Note that harvest in 2004 was higher (180% of the 2000-2004 average) than in any other year. Harvest also declined in 2005 for the E(80 t) and E(110 t) effort scenarios, although only to levels of 126% and 164% of the average 2000-2004 levels (Fig. 6.53D). Trajectories of harvest indicated that, under all MLS scenarios, harvest would gradually increase from levels in 2005 to between 110% and 140% of what it had been between 2000 and 2004 (Fig. 6.53A-C, E). Spatial and seasonal closures (Fig. 6.53A,B) resulted in slightly lower harvest levels than no closures at all (Fig. 6.53E). The combination of seasonal and spatial closures resulted in lower harvest than when spatial or seasonal closures were examined separately (Fig. 6.53C). Increasing the MLS generally resulted in lower harvest under all spatial and seasonal closure scenarios (Fig. 6.53A-C, E). As expected, increasing effort resulted in greater harvest relative to 2000-2004 levels (Fig. 6.53D).

The first set of objectives relating to harvest were about maximising the harvest of smaller fish that fetch a greater price per kilo. Generally, the effects of different management strategies were greater on the proportion of fish <1.2 kg (Fig. 6.55) than the proportion of fish <1.5 kg (Fig. 6.56). Seasonal closures reduced the proportion of fish <1.2 kg (Fig. 6.55) and <1.5 kg (Fig. 6.56) at the middle and end of the projection period (2011-2015 and 2021-2025), although the effect was minimal. Spatial closures had even less effect than seasonal closures, although they resulted in a higher proportion of fish <1.2 kg (Fig. 6.55) and <1.5 kg (Fig. 6.56). Increasing the MLS resulted in a lower proportion of fish <1.2 kg and <1.5 kg, while increasing the effort level resulted in a greater proportion of fish <1.2 kg (Fig. 6.55) and <1.5 kg (Fig. 6.56).

Management strategies involving a MLS of 35 cm, particularly in combination with no spatial closures, resulted in the highest proportion of fish <1.2 kg (Fig. 6.57) and <1.5 kg (Fig. 6.58) compared with status quo management. Management strategies involving a MLS of 40 cm and an effort level of E(80 t) resulted in the lowest proportion of fish <1.2 kg (Fig. 6.57) and <1.5 kg (Fig. 6.58) compared with status quo management.

The first two objectives concerning harvest were that 50% of the harvest should be <1.2 kg (Objective 38) or <1.5 kg (Objective 39) 100% of the time. All management strategies were able to achieve these objectives (Figs. 6.59 and 6.60).
**Fig. 6.54.** Trajectories of mean harvest by Islander commercial fishers relative to mean harvest between 2000 and 2004 averaged over 10 simulations for reefs open to fishing for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
Fig. 6.55. Average (±SE) proportion of available (greater than the minimum legal size) fish less than 1.2kg (2011-2015 black bars and 2021-2025 grey bars).

Fig. 6.56. Average (±SE) proportion of available (greater than the minimum legal size) fish less than 1.5kg (2011-2015 black bars and 2021-2025 grey bars).
**Fig. 6.57.** Average (±SE) proportion of available (greater than the minimum legal size) fish less than 1.2kg (2011-2015 black bars and 2021-2025 grey bars) relative to the proportion of available fish less than 1.2kg under the current management strategy.

**Fig. 6.58.** Average (±SE) proportion of available (greater than the minimum legal size) fish less than 1.5kg (2011-2015 black bars and 2021-2025 grey bars) relative to the proportion of available fish less than 1.5kg under the current management strategy.
Fig. 6.59. Proportion of simulations where the proportion of available (greater than the minimum legal size) fish less than 1.2kg (2011-2015 black bars and 2021-2025 grey bars) exceeds 50% of the harvest (Objective 38) 100% of the time.

Fig. 6.60. Proportion of simulations where the proportion of available (greater than the minimum legal size) fish less than 1.5kg (2011-2015 black bars and 2021-2025 grey bars) exceeds 50% of the harvest (Objective 39) 100% of the time.
The second set of objectives relating to harvest were about achieving a greater harvest in projection years compared with historic years. Seasonal and spatial closures reduced the non-indigenous commercial harvest relative to 2000-2004 levels, although seasonal closures had a greater effect (Fig. 6.61). Increasing the MLS resulted in lower harvest, while increasing the effort level resulted in a greater harvest relative to 2000-2004 levels (Fig. 6.61).

Nearly all management strategies resulted in lower harvest levels compared with status quo management (Fig. 6.62). A MLS of 35 cm and no seasonal closures were the only management strategies to achieve greater harvest compared with status quo management (Fig. 6.62).

The two objectives concerning non-indigenous commercial harvest were that harvest should be greater than 2000-2004 levels (Objective 40) or greater than 95% 2000-2004 levels (Objective 41) 100% of the time. No management strategies were able to achieve either of these objectives (Figs. 6.63 and 6.64). A MLS of 35 cm and no seasonal closures were the management strategies that came closest to achieving these objectives (Fig 6.63 and 6.64).

**Fig. 6.61.** Average (±SE) harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) relative to the average 2000-2004 harvest.
Fig. 6.62. Average (±SE) harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) relative to the non-indigenous harvest under the current management strategy.

Fig. 6.63. Proportion of simulations where the harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds the average 2000-2004 harvest (Objective 40) 100% of the time.
Similar to non-indigenous fishers, seasonal and spatial closures reduced the Islander commercial harvest relative to 2000-2004 levels (Fig. 6.65). Increasing the MLS resulted in lower harvest, while increasing the effort level resulted in a much greater harvest relative to 2000-2004 levels (Fig. 6.65).

Nearly all management strategies concerning spatial and seasonal closures resulted in lower harvest levels compared with status quo management (Fig 6.66). The only exception was when there were no seasonal or spatial closures and a MLS of 35 cm (Fig 6.66). All effort levels resulted in substantially greater harvest than under status quo management (Fig 6.66).

The two objectives concerning Islander commercial harvest were that harvest should be greater than 2000-2004 levels (Objective 42) or greater than 95% 2000-2004 levels (Objective 43) 100% of the time. Only management strategies with no seasonal or spatial closures and management strategies for all effort levels were able to achieve both these objectives (Fig. 6.67 and 6.68).
Fig. 6.65. Average (±SE) harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) relative to the average 2000-2004 harvest.

Fig. 6.66. Average (±SE) harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) relative to the current management strategy.
Fig. 6.67. Proportion of simulations where the harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds the average 2000-2004 harvest (Objective 42) 100% of the time.

Fig. 6.68. Proportion of simulations where the harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 95% of the average 2000-2004 harvest (Objective 43) 100% of the time.
The final objective relating to harvest (Objective 44) was about each sector catching the allocated quota levels (40 t, 55 t, 70 t and 85 t). For non-indigenous commercial fishers, seasonal and spatial closures reduced harvest relative to each quota level, although seasonal closures had a greater effect (Figs. 6.69-6.72). Increasing the MLS resulted in lower harvest, while increasing the effort level resulted in a greater harvest relative to each quota level (Figs. 6.69-6.72).

The performance of management strategies in achieving the objective of non-indigenous fishers catching the quota 100% of the time depended on the quota level (Figs. 6.73-6.76). All management strategies achieved the objective for the 40t quota, except the E(80 t) effort level (Fig 6.73). Similarly, all management strategies achieved the objective for the 55 t quota, except the three lowest effort levels (E(80 t), E(110 t) and E(140 t)) (Fig. 6.74). No spatial or seasonal closures in combination with a MLS of 35 cm or 38 cm were the only two management strategies to achieve this objective for the 70 t quota (Fig. 6.75). No management strategies achieved this objective for the 85 t quota level, although management strategies with no spatial or seasonal closures came closest to achieving this objective 100% of the time (Fig. 6.76).

Fig. 6.69. Average (±SE) harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total non-indigenous harvest of 40 t.
Mean non-indigenous harvest relative to 55 t quota

<table>
<thead>
<tr>
<th>Management Strategy</th>
<th>Seasonal closure</th>
<th>Spatial closure</th>
<th>MLS</th>
<th>Effort</th>
</tr>
</thead>
<tbody>
<tr>
<td>Yes</td>
<td>No</td>
<td>Yes</td>
<td>38cm</td>
<td>2004 Effort level</td>
</tr>
<tr>
<td>No</td>
<td>Yes</td>
<td>No</td>
<td>35cm</td>
<td>E(110)</td>
</tr>
<tr>
<td>Yes</td>
<td>No</td>
<td>Yes</td>
<td>40cm</td>
<td>E(140)</td>
</tr>
<tr>
<td>No</td>
<td>No</td>
<td>No</td>
<td>38cm</td>
<td>E(170)</td>
</tr>
</tbody>
</table>

Fig. 6.70. Average (±SE) harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total non-indigenous harvest of 55 t.

Mean non-indigenous harvest relative to 70 t quota

<table>
<thead>
<tr>
<th>Management Strategy</th>
<th>Seasonal closure</th>
<th>Spatial closure</th>
<th>MLS</th>
<th>Effort</th>
</tr>
</thead>
<tbody>
<tr>
<td>Yes</td>
<td>No</td>
<td>Yes</td>
<td>38cm</td>
<td>2004 Effort level</td>
</tr>
<tr>
<td>No</td>
<td>Yes</td>
<td>No</td>
<td>35cm</td>
<td>E(80)</td>
</tr>
<tr>
<td>Yes</td>
<td>No</td>
<td>Yes</td>
<td>40cm</td>
<td>E(110)</td>
</tr>
<tr>
<td>No</td>
<td>No</td>
<td>No</td>
<td>38cm</td>
<td>E(140)</td>
</tr>
</tbody>
</table>

Fig. 6.71. Average (±SE) harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total non-indigenous harvest of 70 t.
Fig. 6.72. Average (±SE) harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total non-indigenous harvest of 85 t.

Fig. 6.73. Proportion of simulations where the harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 40 t 100% of the time.
Fig. 6.74. Proportion of simulations where the harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 55 t 100% of the time.

Fig. 6.75. Proportion of simulations where the harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 70 t 100% of the time.
Fig. 6.76. Proportion of simulations where the harvest of fish by non-indigenous fishers (2011-2015 black bars and 2021-2025 grey bars) exceeds 85 t 100% of the time.

For Islander commercial fishers, similar to non-indigenous fishers, seasonal and spatial closures reduced harvest relative to each quota level, although seasonal closures had a greater effect (Figs. 6.77-6.80). Increasing the MLS resulted in lower harvest, while increasing the effort level resulted in a greater harvest relative to each quota level (Figs. 6.77-6.80).

No management strategies were able to achieve the objective of Islander fishers catching the quota 100% of the time for any quota level (Fig. 6.81). The highest effort level (E(170 t)) was the management strategy that came closest to achieving this objective for the 40 t quota level (Fig 6.81). No management strategies achieved this objective in any simulations for the E(55 t), E(70 t) and E(85 t) effort levels.
Fig. 6.77. Average (±SE) harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total Islander harvest of 40 t.

Fig. 6.78. Average (±SE) harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total Islander harvest of 55 t.
Fig. 6.79. Average (±SE) harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total Islander harvest of 70 t.

Fig. 6.80. Average (±SE) harvest of fish by Islander fishers (2011-2015 black bars and 2021-2025 grey bars) relative to a total Islander harvest of 85 t.
6.3.6. Sex Ratios

The final stakeholder objective was related to maintaining sex ratios in a ‘healthy’ state. As coral trout are protogynous hermaphrodites, fishing pressure may disproportionately reduce the proportion of males in the population relative to females, which may result in sperm limitation. Therefore, stakeholders were interested in maintaining the proportion of males in projection years at or above current levels (2000-2004). In the projections, trajectories of sex ratios (Male/Female) increased relative to the 2000-2004 sex ratios under all management strategies (Fig. 6.82). The effects of seasonal and spatial closures were similar in increasing the proportion of males relative to 2000-2004 levels (Fig. 6.82A-C). Although the proportion of males increased from 2000-2004 levels with no spatial or seasonal closures (Fig. 6.82E), the proportion of males was lower than with spatial or seasonal closures. Increasing the MLS tended to result in a slightly greater proportion of males relative to 2000-2004 levels, although this varied through time (Fig. 6.82A-C, E). Fishing effort had the greatest effect on sex ratios, with increasing effort levels resulting in a reduced proportion of males relative to 2000-2004 levels (Fig. 6.82D).
Fig. 6.82. Trajectories of mean sex ratio (M/F) relative to mean sex ratio between 2000 and 2004 averaged over 10 simulations for reefs open to fishing for three minimum size limits (38 cm, 35 cm and 40 cm) with A) a spatial closure (10 nm radius round Murray, Darnley, Stephens and Yorke Islands), B) a two month seasonal closure from September 1 to October 31, C) spatial and seasonal closures, D) four effort levels corresponding to total allowable catches of 80 t, 110 t, 140 t and 170 t, and E) no spatial or seasonal closures.
The proportion of males at the middle and end of the projection period (2011-2015 and 2021-2025) relative to the proportion of males in 2000-2004 was greater with seasonal or spatial closures than without closures (Fig. 6.83). Increasing the MLS resulted in a greater proportion of males relative to the proportion of males in 2000-2004, although the effect was less than for seasonal and spatial closures (Fig. 6.83). Increasing the effort level resulted in a lower proportion of males relative to 2000-2004 levels (Fig. 6.83).

Nearly all management strategies resulted in a greater proportion of males compared with status quo management (Fig 6.84). A MLS of 35 cm combined with no seasonal and spatial closures and the E(170t) effort level were the only two management strategies that did not achieve a greater proportion of males compared with status quo management (Fig 6.84).

The specific objective concerning sex ratios was to maintain the proportion of males above the average proportion of males between 2000 and 2004 100% of the time (Objective 45). All management strategies were able to achieve this objective (Fig. 6.85).

Fig. 6.83. Average (±SE) proportion of males (2011-2015 black bars and 2021-2025 grey bars) relative to the average proportion of males in 2000-2004.
**Fig. 6.84.** Average (±SE) proportion of males (2011-2015 black bars and 2021-2025 grey bars) relative to the proportion of males under current management strategy.

**Fig. 6.85.** Proportion of simulations where the proportion of males (2011-2015 black bars and 2021-2025 grey bars) exceeds the proportion of males in 2000-2004 (Objective 45) 100% of the time.
6.3.7 Performance Summary

The results for the major stakeholder objectives are aggregated and synthesised in Table 6.7, which shows the mean value of the performance indicators from each management strategy in the last five years of projection (2021-2025) relative to the reference value across all simulations. The performance indicators relate to the four categories of management objective: conservation, stock, harvest and economic. Care must be taken in interpreting the absolute values of the quantities in Table 6.7. This is because the purpose of evaluating management strategies using an MSE is to make comparisons among strategies, and not to make claims that a particular management strategy will lead to a particular outcome.

The model results show that increasing the MLS is the most effective way of maximising the conservation objective when spatial closures are in place (note that the conservation objective relates to closed reefs only and only certain spatial closures). Reducing effort to levels equivalent to that required to achieve a harvest of 40 t per sector (E(80 t)) was the most effective way to achieve the stock objectives. The management strategy that was best at maximising the harvest objective depended on the sector. Specifically, reducing the MLS to 35 cm and having no seasonal or spatial closures achieves the highest harvest for non-indigenous fishers. Increasing effort to levels equivalent to that required to achieve a harvest of 85t per sector (E(170 t)) was the most effective way to maximise the harvest for Islander fishers. No seasonal closures and a reduced MLS to 35 cm also maximised the harvest for fish less than 1.5 kg. Similar to the harvest objectives, the management strategy that was best at maximising the economic objective depended on the sector. Non-indigenous catch rates were maximised best by reducing effort, while Islander catch rates were maximised with a spawning closure and reduced MLS.
Table 6.7. Summary of performance indicators for different objectives from the last five years of the projection period (2021-2025) under different management strategies. Colours blue: the highest (best) value for a column, green the second highest value, red the lowest (worst) value and orange the second lowest value(MLS, minimum legal size; SB, spawning biomass; AB, available biomass; TB, total biomass; I, Islander; N, non-indigenous; CPUE, catch per unit of effort).

<table>
<thead>
<tr>
<th>Strategy</th>
<th>Seasonal Closure</th>
<th>Spatial Closure</th>
<th>MLS</th>
<th>Effort</th>
<th>Objective</th>
<th>Stock</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Conservation</td>
<td>AB on open reefs (unfished)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>SB on closed reefs (00-04)</td>
<td></td>
</tr>
<tr>
<td>Sep-Oct</td>
<td>10 nm</td>
<td>35 cm</td>
<td>2004</td>
<td></td>
<td>0.89</td>
<td>1.46</td>
</tr>
<tr>
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<td>0.87</td>
<td>1.29</td>
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<td></td>
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</tr>
<tr>
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<td>38 cm</td>
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<td>1.58</td>
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<tr>
<td>None</td>
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<td>E(110 t)</td>
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</tr>
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</tr>
<tr>
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<td>38 cm</td>
<td>E(170 t)</td>
<td></td>
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</tr>
</tbody>
</table>
6.4. Discussion

Management of the ETS RLF is complex, not only because there are two distinct commercial sectors often with disparate objectives, but also because there is a range of management strategies currently in place, and numerous other management strategies proposed, such as seasonal and spatial closures, minimum legal sizes and limits on catch. However, the potential effects of these management strategies on coral trout stocks and on the harvest patterns for coral trout in Torres Strait are unknown. Therefore, there is a need to develop ways to address the effectiveness of alternative management strategies in the ETS RLF. This project provided a formal context within which to evaluate a range of alternative management strategies, related to the harvest and conservation of coral trout, the major target species of the ETS RLF.

6.4.1. Management Strategies

Seasonal closures

Seasonal closures were implemented on the GBR in 2004 with the aim to protect spawning stocks of coral trout and other coral reef fish. Currently, there are no seasonal closures in the ETS RLF, although the potential to introduce such closures to complement management arrangements on the GBR has been discussed among fishery stakeholders. In this chapter, the effect of seasonal closures was addressed in the context of simply removing effort from the fishery for two months during the peak spawning times for coral trout in Torres Strait (Chapter 3). The importance of such seasonal closures is increased if the catchability of coral trout increases during the closure. However, previous analyses on the GBR (Mapstone et al. 2001) have shown that catchability of coral trout does not increase significantly during the spawning season.

In general, seasonal closures had conservation, stock and economic benefits, but tended to reduce the ability to satisfy management objectives related to harvest. Seasonal closures had a greater impact on the harvest for non-indigenous fishers than for Islander fishers, but the relative effects of spawning closures on CPUE was similar between sectors. The absence of seasonal closures had a similar effect to increasing effort, although it was somewhat muted because the amount by which effort was increased through removing seasonal closures was less than the difference in effort between the different effort levels examined. Further research is being planned to develop the ability to evaluate the potential effects of spawning disruptions caused by line fishing, in an effort to judge the potential efficacy of seasonal closures in protecting the spawning stock.

Spatial closures

Areas have been closed to fishing on the GBR for many decades and, in 2004, the amount of area closed to fishing on the GBR increased significantly through a new zoning plan based on protecting a minimum percentage of all representative areas. In Torres Strait, there have been no areas closed to fishing within the area in which the ETS RLF operates. However, some stakeholders have expressed support for the closure of some areas in Torres Strait to fishing. In particular, Torres Strait Islanders have articulated a desire for closures to non-indigenous fishing within a 10 nm radius around some of the ETS islands. In this chapter, the effects of spatial closures were addressed by excluding fishing by both sectors within a 10 nm radius around Murray, Darnley, Yorke and Stephens Island. Restricting fishing by both sectors within this area, rather than only restricting non-indigenous fishing, maximised the contrast
between spatial closures and no closures, such that the effects of closures are exaggerated.

It was not possible to assess the direct effects of spatial closures on the conservation objective (spawning biomass on closed reefs), as there were no areas closed to fishing in any other management strategies with which to compare performance. The effects of area closures varied among the three stock objectives. Area closures tended to reduce available biomass on open reefs but increase total biomass and proportion of males on all reefs. This is most likely because the latter two objectives include all reefs, including those closed to fishing, while the available biomass objective was restricted to open reefs only. Area closures tended to reduce the ability to satisfy management objectives related to harvest and economics for both non-indigenous and Islander fishers. This is simply due to the reduction in the amount of area available to the fishery and, therefore, the amount of available biomass. These results are qualitatively similar to those reported for coral trout on the GBR (Mapstone et al. 2004, Little et al. 2007a)

Minimum Legal Size

Currently, there is a minimum legal size of 38 cm for common coral trout in Torres Strait. This minimum size allows coral trout to spawn at least once, and potentially more, before becoming vulnerable to harvest. In this chapter, we examined the effects of either increasing or decreasing this limit. Results showed that increasing the minimum legal size gave greater conservation benefits, but tended to reduce the ability to satisfy management objectives related to harvest and economics. Interestingly, the effects of changing the minimum legal size varied among stock objectives. An increase in the minimum legal size tended to increase the total biomass and proportion of males on all reefs. However, it is important to note that it is invalid to compare the effect of different MLS on available biomass, due to the change in definition of available biomass under different MLS.

Fishing Effort

Before this project commenced, the commercial sectors of the ETS RLF were not managed by any effort or catch limits. During the course of this project, steps have been made to move the ETS RLF towards a quota managed fishery with a Total Allowable Commercial Catch (TACC) for coral trout and a 50:50 quota split between non-indigenous and Islander fishers. In this chapter, we used effort as a proxy for catch (based on sector-specific average catch rates), as ELFSim does not currently have the capacity to directly simulate catch quotas. Five different effort levels were examined; the first corresponded to effort reported in 2004, and the other four effort levels represented the estimated effort required to harvest 80 t, 110 t, 140 t and 170 t (across both sectors). The different levels of TAC were based on a range of potential TACCs for future management of the fishery (e.g. under a TAC of 80 t, each sector would be allocated 40 t). For the non-indigenous sector, the 2004 effort level (3007 dory days) was greater than the estimated effort required to harvest the highest quota of 85 t (2342 dory days). In contrast, the 2004 effort level for the Islander sector (1545 dory days) was lower than the effort required to harvest the lowest quota of 40 t (2292 dory days). This was mainly due to the fact that non-indigenous fishers had historically harvested significantly more coral trout than Islander fishers, but also because historic catch rates for Islander fishers were substantially lower than those for non-indigenous fishers (Chapter 3). It is important to factor these differences into the interpretation of the effects of effort on stakeholder objectives.

In general, changes in effort had a large effect on the ability to achieve management objectives, as found previously on the GBR (Mapstone et al. 2004, Little et al. 2007).
Not surprisingly, results showed that increasing effort had benefits for harvest, but tended to reduce the ability to satisfy management objectives related to the stock and economics. It is important to take care in interpreting the effects of the different effort levels in achieving the objectives relating to retaining the quota. Because effort is kept constant through projection years under each effort level, achieving the corresponding quota (e.g. achieving the 40 t quota under the E(80 t) effort level) is highly unlikely to be maximised, as clearly higher effort levels will achieve a greater catch.

Interestingly, Islander fishers were not able to retain any of the three larger quota levels (55 t, 70 t, 85 t) under any management strategy, including increasing effort to more than three times the 2004 level. This is likely due to two factors; i) the available fishing area for Islander fishers is restricted to a relatively small area around their home island (Chapter 3) and, therefore, significant increases in effort within this area are likely to lead to local depletions, and ii) Islander fishers have much lower catch rates than non-indigenous fishers (Chapter 3). This suggests that if the Islander commercial sector is to be able to catch their quota, they will need to modify existing fishing practices significantly to reach more remote fishing areas and improve catch rates.

6.4.2. Model Issues

Spawning biomass levels in the closed areas did not reach pre-exploitation levels for a significant amount of the time, and the objectives of achieving spawning biomass in the closed areas above 90% unfished levels 100% of the time could not be achieved by any of the management strategies that were explored. There were several reasons for this. First, it is unlikely that even an unexploited population would maintain its status above the carrying capacity for a majority of the time. Second, while the areas open to fishing might benefit from the areas closed to fishing in subsidising reproduction, this comes at a slight cost to the populations on the closed reefs, making it less likely that the populations in the protected areas make a full recovery to pre-exploitation levels. Third, infringement, which may especially affect relatively small area closures (Little et al. 2005), was also applied to the closed reefs at a rate of 5% of the reef's attractiveness. Thus, areas that were putatively closed to fishing were actually subjected to some fishing pressure.

It is important to note that the biological parameters in these simulations were based on a single species, common coral trout. There are four species of coral trout harvested in Torres Strait (Chapter 3), although common coral trout account for approximately 80% of this harvest. Because the proportion of each species in catch by reef was unknown, and the complexities of modelling four species, it was necessary to assume that all species of coral trout harvested shared the biological characteristics of common coral trout. As we have demonstrated that there are differences in biology among species of coral trout in Torres Strait (Chapter 3), assessing the validity of this assumption should be a priority area for future research.

Furthermore, the model was based on the biology of common coral trout from the GBR, and not on the known biology of this species in Torres Strait presented in Chapter 3. This approach was necessary, as the biological parameters estimated for common coral trout in Torres Strait (Chapter 3) were highly likely to be biased, as there was a lack of smaller and younger fish sampled, which are essential for reliable estimates of growth and maturity. Obtaining these smaller samples and re-estimating biological parameters should be a priority area for future research.

Although stakeholders were interested in using the 2000 to 2004 period as a reference point for a number of objectives, the highly variable catch and effort data...
for both sectors in this period (Chapter 3) introduce uncertainty into the interpretation of the results. For example, the average catch rate for the non-indigenous sector between 2000 and 2004 was much higher than in previous years. Therefore, the objective of achieving or exceeding the 2000-2004 catch rates in projection years was rarely achieved. Clearly there is a need to validate catch and effort records from commercial logbooks to ensure the data used in simulations is reliable.

Finally, the effort model used in the simulations relied on fleet aggregate behaviour and assigned non-indigenous commercial fishing effort based on historical reef catch rates. Although a complex vessel dynamics model consisting of individual fishing vessels operating at a daily time step has been developed in ELFSim for use on the GBR, we are confident that the approach taken in this study was robust given that Little et al. (2007a) demonstrated that there was minimal difference in performance between the two effort models. Islander commercial effort, however, was modelled on an even coarser spatial scale than non-indigenous effort, and did not consider annual or seasonal variation (i.e. static). Variation in Islander commercial effort across space or time may affect the model outputs. Future research should focus on obtaining greater resolution in the effort data for the Islander commercial sector.

6.4.3. Summary

This research has built upon the simulation model ELFSim that has been the basis for MSE on the GBR. We have applied the ELFSim model to the unique features of the ETS RLF which, unlike the GBR, has two commercial sectors. In consultation with stakeholders, management objectives and strategies for the ETS RLF were evaluated in ELFSim. A seasonal closure was a good measure for addressing conservation, stock and economic objectives, although the closure was essentially removal of effort, and did not consider the displacement of effort to other times of the year. Closing areas to fishing and increasing the MLS were generally good strategies for stock objectives, but performed poorly for harvest and economic objectives. Lowering effort was the more robust strategy for achieving stock and economic objectives. Trade-offs among conflicting management objectives were highlighted by the effect that although reducing effort increases the chances of achieving these objectives, it reduces the likelihood of achieving the harvest objective.
7. Research and Monitoring Recommendations

In summary, we provide the following recommendations for future research and monitoring of the ETS RLF to improve fishery and biological knowledge and further develop the MSE:

1. Need to develop a long term monitoring program that provides a comprehensive and structured approach to the collection of appropriate age or length-structured data for the main target species from both the non-indigenous, Islander and traditional sectors.

2. Need for improved reporting in the compulsory commercial logbooks and Islander docket books. Reporting of catch in both numbers and weight needs to be more consistent and comprehensive. Fishers need to be encouraged to fill out logbooks in their entirety. Logbook data should be compared to unload/buyer dockets for validation and data checking.

3. Need for a better measure of effort in the commercial logbooks and Islander docket books to provide a more reliable indicator of CPUE, and in turn, stock abundance. Fishers should be encouraged to record search and fishing times, number of fishers, and days when zero catches occurred.

4. Need to obtain more reliable estimates of biological parameters for the target species in the ETS RLF such as the coral trout species and barramundi cod. Specifically, there is a need to collect small and young individuals to provide more reliable estimates of growth and maturity. There is also the need to determine whether spawning of coral trout occurs in other months of the year. Specifically, samples need to be collected during the months December through March.

5. Need to consider specific management arrangements for passionfruit trout, *Plectromomus areolatus*, such as an increase in the current minimum legal size or the introduction of a maximum legal size due to the larger size at sex change compared with other coral trout species.

6. Need to obtain a better description of the reefs and shoal areas that are fished by Indigenous fishers to provide a clearer picture of the total area in which Islanders are able to fish.

7. Need to obtain reliable estimates of subsistence harvest of reef fish to include in future assessments of the fishery.

8. Need to update the MSE once some of the needs listed above have been fulfilled and additional information is available. This may require determining whether stakeholder objectives have changed since the initial model run.

9. Need to obtain a better understanding of the source-sink relationships between reefs, larval dispersal, self-seeding and larval subsidy and to determine the sensitivity of management strategies to different models of larval migration.
8. References


Great Barrier Reef food fish stocks. Fisheries Research and Development Corporation, Canberra.


commercial demersal reef line fishery. FRDC Report No. 96/138, Fisheries Research and Development Corporation, Canberra.


Appendix 1: Voluntary research logbooks for Islander commercial fishers

*Number of each species category caught by Murray Islander fishers*

| Date | Fisher | Common coral trout | Leopard or passion-fruit trout | Island or Bar-cheeked trout | Blue-spot trout | Footballer trout | Coronation trout | Maori wrasse | Barra cod | Small cod | Large cod | Red Emperor |
|------|--------|--------------------|--------------------------------|-----------------------------|----------------|-----------------|-----------------|--------------|----------|----------|-----------|-----------|-------------|
|      |        |                    |                                |                             |                |                 |                 |              |          |          |           |           |             |
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|      |        |                    |                                |                             |                |                 |                 |              |          |          |           |           |             |
|      |        |                    |                                |                             |                |                 |                 |              |          |          |           |           |             |
### Number of each species of Mixed Reef fish category caught by Murray Islander fishers

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<thead>
<tr>
<th>Date</th>
<th>Fisher</th>
<th>Spanish Mack</th>
<th>Salmon Mack</th>
<th>Trevally</th>
<th>Stripey Bass</th>
<th>Red Bass</th>
<th>Yellow tail Emperor</th>
<th>Spangled Emperor</th>
<th>Tusk Fish</th>
<th>Pink Eared Emperor</th>
<th>Longnose Emperor</th>
<th>Other Emperors</th>
<th>Other Snappers</th>
<th>All Others</th>
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Appendix 2: Observer survey data sheets for commercial Islander and non-indigenous fishers

F&F – ETS Reef Line Project

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<th>Reef</th>
<th>Fishing</th>
<th>Hang notes</th>
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<th>Depth (m)</th>
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Fate: R = released, A = afloat (fish floated), H = kept as hard bait, B = signs of barotrauma, V = vented
## Day Data Sheet

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<tr>
<th>Fisher Name</th>
<th>Session</th>
<th>Coral Trout</th>
<th>Barramundi Cod</th>
<th>Spanish Mackerel</th>
<th>Red Emperor</th>
<th>Maori Wrasse</th>
<th>Other</th>
<th>Other</th>
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Please return to: Cameron Murchie, CRC Reef Research Centre, James Cook University, Townsville, 4811. Telephone: 07 47814158 Fax: 07 47814099.
## Appendix 3: Species List

List of species caught by Islander fishers (I) or non-indigenous fishers (N) in the eastern Torres Strait reef line fishery. Catch refers to species that were caught and bycatch refers to species that were never harvested. Species group refers to group in which each species was placed for analysis of species composition.

<table>
<thead>
<tr>
<th>Family</th>
<th>Species</th>
<th>Common Name</th>
<th>Catch</th>
<th>Bycatch</th>
<th>Species Group</th>
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<tbody>
<tr>
<td>Balistidae</td>
<td>Unknown</td>
<td>Trigger fish</td>
<td>N</td>
<td>N</td>
<td>NA</td>
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<tr>
<td>Belonidae</td>
<td><em>Tylosurus crocodilus</em></td>
<td>Crocodile Longtom</td>
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<td>Caesionidae</td>
<td><em>Caesio teres</em></td>
<td>Gold Banded Fusilier</td>
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<td>Carangidae</td>
<td><em>Elagatis bipinnulata</em></td>
<td>Rainbow runner</td>
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<td>I</td>
<td>Trevallies</td>
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<tr>
<td>Carangidae</td>
<td>Unknown</td>
<td>Trevally (All species)</td>
<td>I, N</td>
<td>-</td>
<td>Trevallies</td>
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<tr>
<td>Carcharhinidae</td>
<td><em>Carcharhinus ambyrhythchos</em></td>
<td>Grey Reef Shark</td>
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<td>I, N</td>
<td>Sharks</td>
</tr>
<tr>
<td>Carcharhinidae</td>
<td>Unknown</td>
<td>Shark</td>
<td>I, N</td>
<td>I, N</td>
<td>Sharks</td>
</tr>
<tr>
<td>Hemigaleida</td>
<td><em>Triaenodon obesus</em></td>
<td>White-tip Shark</td>
<td>I, N</td>
<td>I, N</td>
<td>Sharks</td>
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<td>Labridae</td>
<td><em>Cheilinus undulatus</em></td>
<td>Maori Wrasse</td>
<td>I, N</td>
<td>I, N</td>
<td>Wrasses</td>
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<tr>
<td>Labridae</td>
<td><em>Choerodon cyanodus</em></td>
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*Fishing & Fisheries Technical Report No. 1*
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*Individuals of these species were released as they were less than the minimum legal size, but these species (except Maori wrasse which is now a no-take species) are likely to be harvested when above the minimum legal size.*

**Individuals of these species were recorded in research logbooks as harvested by Islander fishers.*
Appendix 4: Intellectual Property

No patentable or marketable products or processes have arisen from this research. All results will be published in scientific and non-technical literature. The raw data from compulsory fishing logbooks remains the intellectual property of the Queensland Department of Primary Industries and Fisheries and the Australian Fisheries Management Authority. Islander catch data remains the intellectual property of the respective Island Council. Raw catch data provided by individual fishers to project staff remains the intellectual property of the fishers. With the exception of background intellectual property described above, the intellectual property accruing from the analysis and interpretation of data in this report belongs to the Reef and Rainforest Research Centre, which has agreed to licence its use to all members of the CRC Reef and Torres Strait.

Appendix 5: Staff

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Sara Busilacchi

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**Liaison Officers:** Annabel Jones  
Ann Penny

**Administrative Officers:** Iesha Stewart  
Katia Bazaka