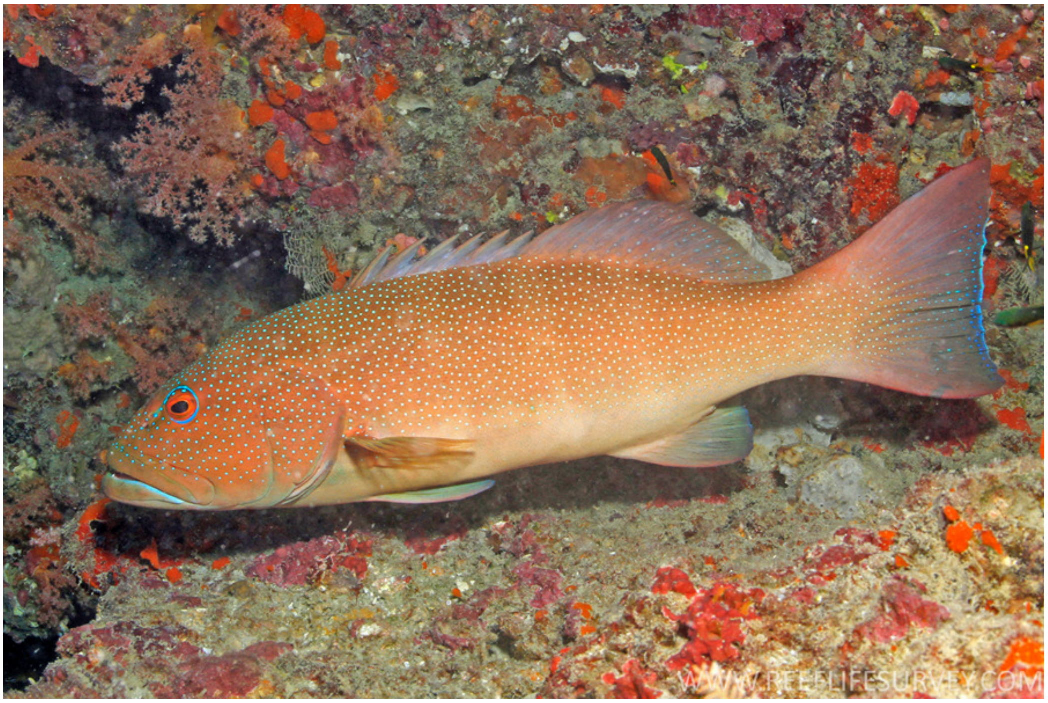
Stock assessment of the Queensland east coast common coral trout (*Plectropomus leopardus*) fishery



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This publication provides an assessment of the state of the population of common coral trout, one of Australia's important commercial and recreational coral reef fish, with recommendations for management, future research and data collection.

Cover photo: *Plectropomus leopardus* on the Great Barrier Reef (source: Graham Edgar, www.reeflifesurvey.com, Creative Commons by Attribution licence for non-commercial use).

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Executive summary

Common coral trout *Plectropomus leopardus* is an iconic fish of the Great Barrier Reef (GBR) and is the most important fish for the commercial fishery there. Most of the catch is exported live to Asia.

This stock assessment was undertaken in response to falls in catch sizes and catch rates in recent years, in order to gauge the status of the stock. It is the first stock assessment ever conducted of coral trout on the GBR, and brings together a multitude of different data sources for the first time.

The GBR is very large and was divided into a regional structure based on the Bioregions defined by expert committees appointed by the Great Barrier Reef Marine Park Authority (GBRMPA) as part of the 2004 rezoning of the GBR. The regional structure consists of six Regions, from the Far Northern Region in the north to the Swains and Capricorn–Bunker Regions in the south. Regions also closely follow the boundaries between Bioregions. Two of the northern Regions are split into Subregions on the basis of potential changes in fishing intensity between the Subregions; there are nine Subregions altogether, which include four Regions that are not split. Bioregions are split into Subbioregions along the Subregion boundaries. Finally, each Subbioregion is split into a "blue" population which is open to fishing and a "green" population which is closed to fishing.

The fishery is unusual in that catch rates as an indicator of abundance of coral trout are heavily influenced by tropical cyclones. After a major cyclone, catch rates fall for two to three years, and rebound after that. This effect is well correlated with the times of occurrence of cyclones, and usually occurs in the same month that the cyclone strikes. However, statistical analyses correlating catch rates with cyclone wind energy did not provide significantly different catch rate trends. Alternative indicators of cyclone strength may explain more of the catch rate decline, and future work should investigate this.

Another feature of catch rates is the phenomenon of social learning in coral trout populations, whereby when a population of coral trout is fished, individuals quickly learn not to take bait. Then the catch rate falls sharply even when the population size is still high. The social learning may take place by fish directly observing their fellows being hooked, or perhaps heeding a chemo-sensory cue emitted by fish that are hooked. As part of the assessment, analysis of data from replenishment closures of Boult Reef in the Capricorn–Bunker Region (closed 1983–86) and Bramble Reef in the Townsville Subregion (closed 1992–95) estimated a strong social learning effect.

A major data source for the stock assessment was the large collection of underwater visual survey (UVS) data collected by divers who counted the coral trout that they sighted. This allowed estimation of the density of coral trout in the different Bioregions (expressed as a number of fish per hectare). Combined with mapping data of all the 3000 or so reefs making up the GBR, the UVS results provided direct estimates of the population size in each Subbioregion.

A regional population dynamic model was developed to account for the intricacies of coral trout population dynamics and catch rates. Because the statistical analysis of catch rates did not attribute much of the decline to tropical cyclones, (and thereby implied "real" declines in biomass), and because in contrast the UVS data indicate relatively stable population sizes, model outputs were unduly influenced by the unlikely hypothesis that falling catch rates are real. The alternative hypothesis that UVS data are closer to the mark and declining catch rates are an artefact of spurious (e.g., cyclone impact) effects is much more probable.

Judging by the population size estimates provided by the UVS data, there is no biological problem with the status of coral trout stocks. The estimate of the total number of *Plectropomus leopardus* on blue zones on the GBR in the mid-1980s (the time of the major UVS series) was 5.34 million legal-sized fish, or about 8400 t exploitable biomass, with an additional 3350 t in green zones (using the current zoning which was introduced on 1 July 2004). For the offshore regions favoured by commercial fishers, the figure was about 4.90 million legal-sized fish in blue zones, or about 7700 t exploitable biomass.

There is, however, an economic problem, as indicated by relatively low catch rates and anecdotal information provided by commercial fishers. The costs of fishing the GBR by hook and line (the only method compatible with the GBR's high conservation status) are high, and commercial fishers are unable to operate profitably when catch rates are depressed (e.g., from a tropical cyclone). The economic problem is compounded by the effect of social learning in coral trout, whereby catch rates fall rapidly if fishers keep returning to the same fishing locations. In response, commercial fishers tend to spread out over the GBR, including the Far Northern and Swains Regions which are far from port and incur higher travel costs.

The economic problem provides some logic to a reduction in the TACC. Such a reduction during good times, such as when the fishery is rebounding after a major tropical cyclone, could provide a net benefit to the fishery, as it would provide a margin of stock safety and make the fishery more economically robust by providing higher catch rates during subsequent periods of depressed catches. During hard times when catch rates are low (e.g., shortly after a major tropical cyclone), a change to the TACC would have little effect as even a reduced TACC would not come close to being filled. Quota adjustments based on catch rates should take account of long-term trends in order to mitigate variability and cyclone effects in data.

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Acknowledgements

Many scientists and fishing industry participants were extremely generous with their time, data and other products of their work during the course of this assessment. Their contributions have resulted in an extremely valuable collection of data and information that is unique in Australian fisheries science.

Foremost acknowledgement must go to Darren Cameron of the Great Barrier Reef Marine Park Authority (GBMRPA), who went well beyond the call of duty to help in any way he could with the conduct of the stock assessment. Among many other things, he helped to arrange interviews with as wide a cross-section of commercial fishers on the Great Barrier Reef as he could find, and with Tony and Avril Ayling, the divers who have conducted many of the underwater visual surveys of the Reef over a period of 30 years. The Great Barrier Reef Marine Park Authority was also extremely helpful with the provision of reports, articles, maps, mapping data and data from underwater visual surveys of the Great Barrier Reef.

Many commercial fishers freely gave their time in interviews to express their knowledge and opinions about the fishery, without which the methodology and results of this assessment would have been incorrect. We are very grateful to them.

Tony and Avril Ayling were very generous with their knowledge and were able to provide data from underwater visual surveys that we had not been able to locate anywhere else.

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Dr Hugh Sweatman of the Australian Institute of Marine Science (AIMS) generously provided the underwater visual survey data from the AIMS Long Term Monitoring Project.

Dr Mike Cappo of AIMS freely discussed his research on potential inter-reef habitat for reef fish.

Dr Rich Little of CSIRO, a participant in the original ELF Project and the Principal Investigator in the newer related FRDC project (no. 2011/030), freely gave his ideas and the spatial data that were used in the ELFSim software product spawned by these projects.

Dr Garry Russ of James Cook University provided the age-frequency data collected by his group in the early 1990s which constituted the earliest available age-frequency data for coral trout on the Great Barrier Reef, and coral trout abundance data from the Bramble Reef replenishment experiment for which the reef was closed to fishing from 1992 to 1995.

Jim Higgs and Dr Campbell Davies, both formerly of James Cook University, provided additional data and reports on the Bramble Reef experiment, including unpublished manuscripts.

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Dr David Williamson of James Cook University provided data on the species composition of coral trout on inshore reefs.

Dr André Punt of CSIRO and the University of Washington provided the source code for the Cabezon stock assessment model (written by himself and others), on which the population dynamic model in this assessment was built.

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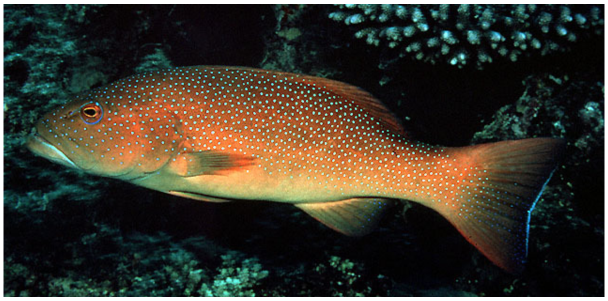
Dr Brigid Kerrigan and Dr John Kung of Fisheries Queensland facilitated meetings with fishery stakeholders, which provided opportunities for valuable discussion of many aspects of the stock assessment.

1. Introduction

1.1 Overview

The common coral trout *Plectropomus leopardus* is the primary target species of the commercial Coral Reef Fin Fish Fishery on Queensland's East Coast, and is an attractive fish for recreational line fishers who can travel to the offshore reefs of the Great Barrier Reef.



Plectropomus leopardus (source: Randall, 1997)

Commercial fishers experienced lower than average catch rates for this species between 2009 and 2012, which added to existing concerns about the level of fishing.

The fishery is located in the Great Barrier Reef Marine Park, established in 1975 and declared a World Heritage Area in 1981. It is a line-only fishery, except for a small amount of recreational spear fishing. Fishing gear such as longlines, nets, traps, dynamite and cyanide, which are used in coral reef fisheries in other parts of the world, are not permitted in this fishery. Commercial fishers generally choose to fish with single hooks and one fisher per vessel or dory. A primary vessel often supports multiple dories.

The fishery is data-rich by southern hemisphere standards. Underwater visual surveys of fish stocks by divers have been conducted on hundreds of reefs since the mid-1980s; large-scale scientific experiments have been conducted on the effects of fishing; and there is extensive literature on the biology of the species. The commercial fishery logbook database began in 1988, and records daily catches to six-nautical-mile resolution, although most records from early years have only 30-nautical-mile resolution.

A complicating factor in the fishery is that fishers' catch rates are affected by changes in fish behaviour and often do not correlate well to abundance of the fish. This point was noted by Beinssen (1989) and has been confirmed many times since.

This stock assessment was commissioned to assess the sustainability of the fishery, estimate the biomass of common coral trout present on the Great Barrier Reef, and estimate the annual yields that can be taken while maintaining specified levels of fish stocks and catch rates.

1.2 The Great Barrier Reef

The Great Barrier Reef (GBR) is the world's largest coral reef system. It contains about 3000 reefs and runs for 2300 km parallel to the Queensland East Coast (GBRMPA, 2009) (see Figure 1).

The GBR is young, both in geological terms and compared to other coral reef systems in the world (Hopley et al., 2007, sections 2.5–2.7). The time of foundation is thought to have been about 400,000 years ago. Recent growth of the GBR has been governed by changes in sea level since the end of the last ice age. Sea level rose from about 125 m below current level about 19,000 years ago, to reach its current level about 6500 years ago (Lambeck et al., 2002; Hopley et al., 2007, section 3.4). The reefs are still evolving, proceeding roughly along the cycle depicted in Figure 2 (Hopley et al., 2007, Figure 5.7 and sections 8.4–8.6).

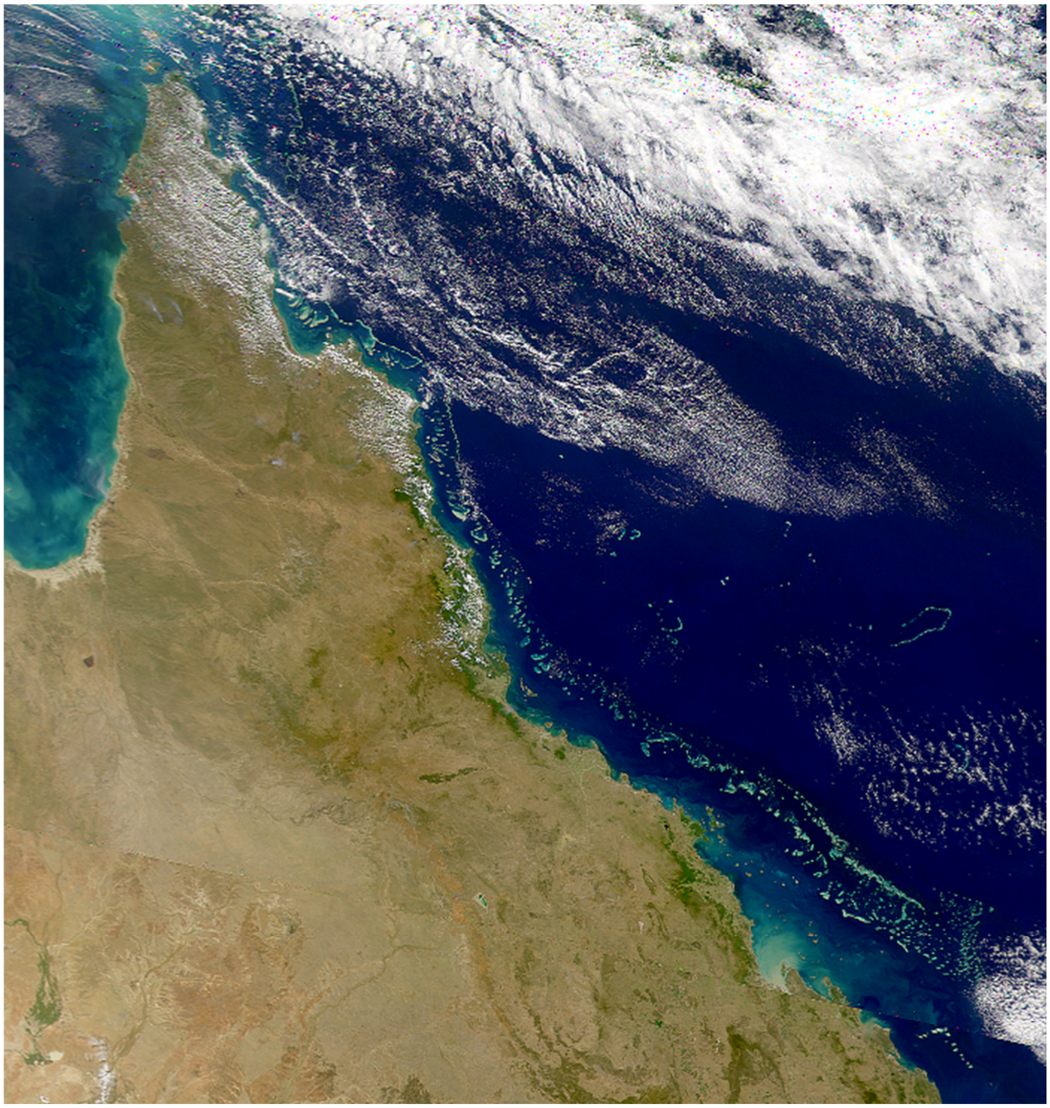


Figure 1: The Great Barrier Reef along the Queensland east coast, viewed from space. Source: SeaWiFS Project, NASA/ Goddard Space Flight Center, and ORBIMAGE, http://visibleearth.nasa.gov, licence similar to Creative Commons by Attribution.

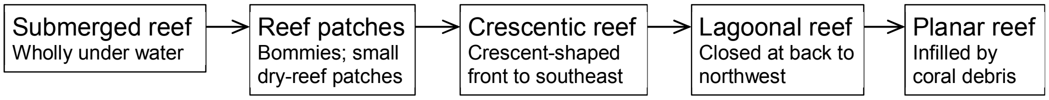


Figure 2: Stages of evolution of coral reefs on the Great Barrier Reef.

The different types of reef are illustrated in Figure 3 and Figure 4. The back reef slope is the habitat on which coral trout are most abundant, followed closely by the front reef slope (Ayling and Ayling, 2000). Reef patches have a lower density of fish but a much greater area than the reef slope, and can be fished in a greater range of tide and weather conditions; hence they form very important fishing grounds (commercial reef fishers, personal communications, 2012). A few fishers specialise in fishing deep submerged reef habitat (30–50 m deep).

It is notable that, somewhat contrary to intuition, reefs in the younger stages of evolution tend to be the more productive habitats for fish. Planar reefs, the most advanced stage, are the least productive: they have no habitat in the interior and often provide only a very narrow strip of reef slope as the only viable shallow-water habitat.

The morphology of each reef in the GBR that had been mapped by that time was classified by Hopley et al. in the early 1980s, from data used in the previous edition of their book (Hopley et al., 2007). This gave rise to the map of reef morphology and tidal ranges reproduced in Figure 5. The southern part of the GBR has the biggest tidal ranges, which can be expected to produce higher flows of nutrients. Lagoonal reefs (with large submerged parts) are most abundant there, as are coral trout (Ayling and Ayling, 2000). Coral reefs are well adapted to strong tidal currents but can be damaged by extreme events such as tropical cyclones.

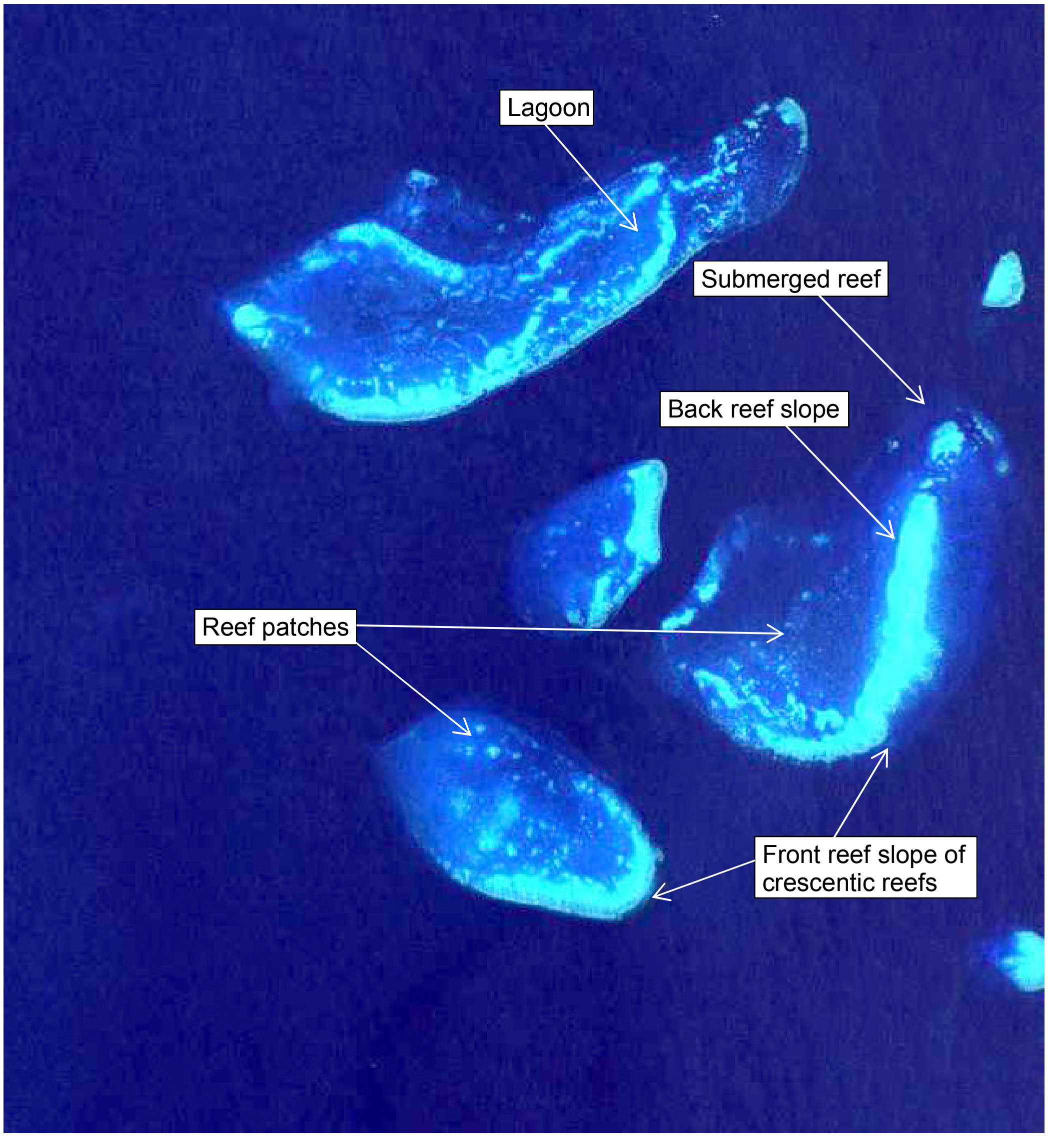


Figure 3: Satellite photograph of a cluster of reefs near Townsville (Bramble Reef, Trunk Reef, Walker Reef and Britomart Reef), showing the different types of reef that form habitat for coral trout. Many productive submerged parts of reefs are not visible on satellite images. Source: NASA, www.dvidshub.net, licence similar to Creative Commons by Attribution.

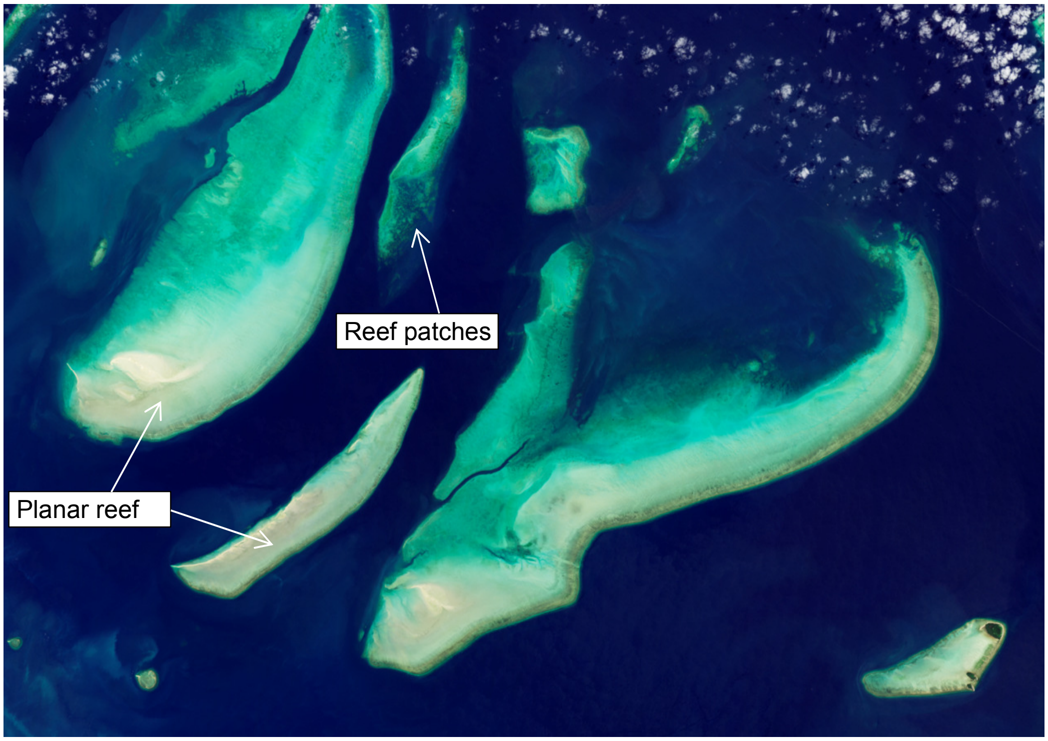


Figure 4: Satellite photograph of a cluster of reefs in Princess Charlotte Bay in the far north of the Great Barrier Reef (Clack Island, Corbett Reef, Grub Reef, Hedge Reef and unnamed reefs 13-123, 13-124 and 13-122), showing planar reefs (the most advanced stage of reef evolution, in which the reef interior is filled with coral debris carried in by waves), which are common in that region. Source: Earth Observatory images by Jesse Allen and Robert Simmon, using data provided by the U.S. Geological Survey, http://visibleearth.nasa.gov, licence similar to Creative Commons by Attribution.

The Great Barrier Reef Marine Park Authority (GBRMPA) introduced the Representative Areas Program (RAP) in 2004, a major part of which was the declaration of many new marine protected areas ("green zones") which were closed to fishing. The green zones were designed to protect biodiversity, but they also afford protection to reef-fish spawning stocks.

As part of the process for determining the Representative Areas, committees of expert scientists divided the GBR into Bioregions. Bioregions were intended to group together neighbouring reefs that had similar characteristics. The resulting Bioregions are shown in Figure 6.

For the purpose of this stock assessment, the GBRMPA Bioregions were used to divide the GBR into six different Regions: the Far Northern Region, the Cooktown Region, the Cairns–Townsville Region, the Mackay Region, the Swains Region, and the Capricorn–Bunker Region. Because fishing intensity increased from north to south in the Far Northern and Cairns–Townsville Regions, these were subdivided into three and two Subregions respectively. The Far Northern Region was divided into the Cape York Subregion, the Lockhart River Subregion, and the Princess Charlotte Bay Subregion; and the Cairns–Townsville Region was divided into the Cairns Subregion and the Townsville Subregion. The Regions and Subregions are shown in map form in Figure 7. The assessment also uses the original Bioregions, split from north to south into Subbioregions where necessary to fit into the Subregions.

The use of the Regions, Subregions and Subbioregions in the coral trout population model is described in detail in Chapter 6.

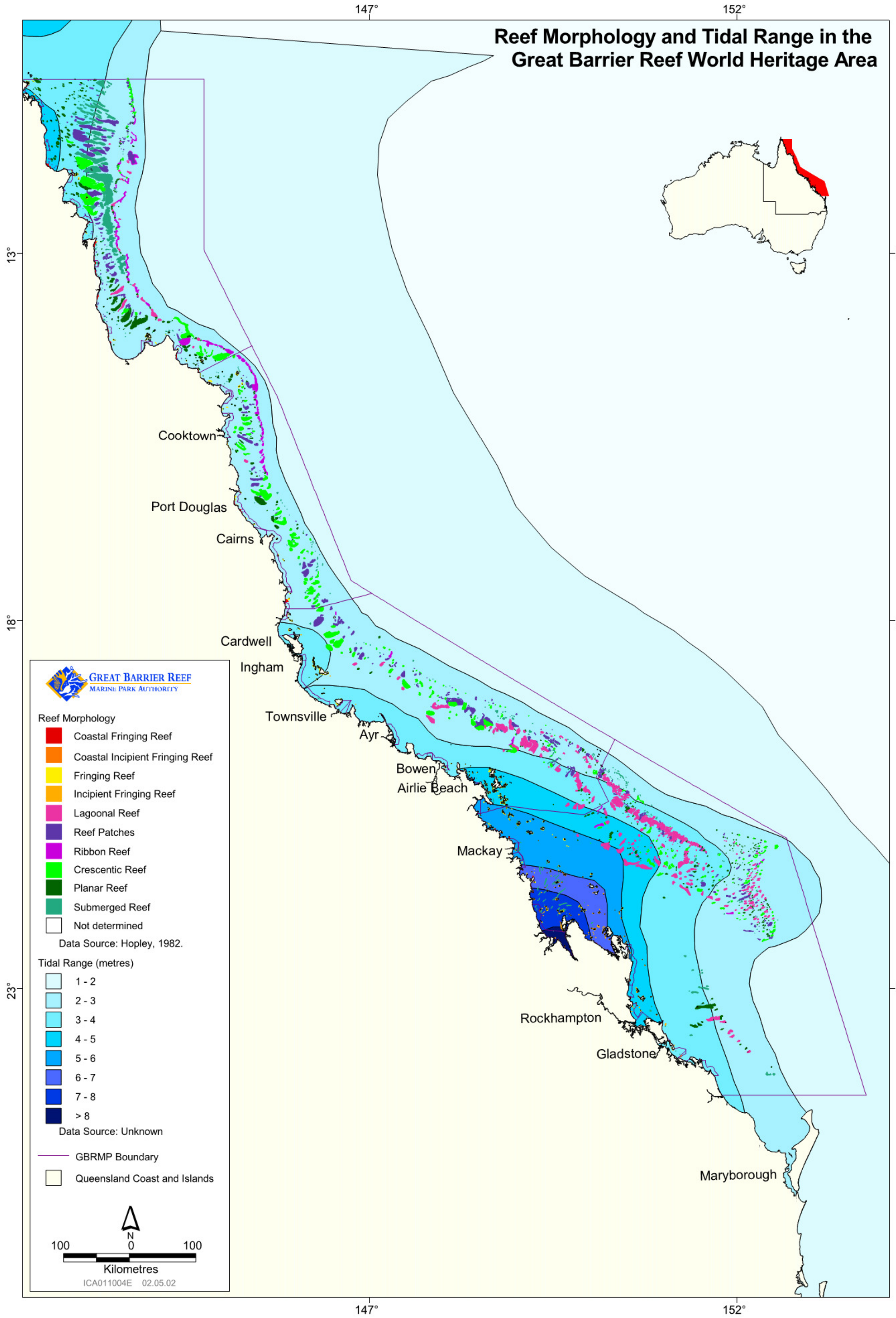


Figure 5: Reef morphology and tidal ranges of the Great Barrier Reef. Source: Great Barrier Reef Marine Park Authority, www.gbrmpa.gov.au.

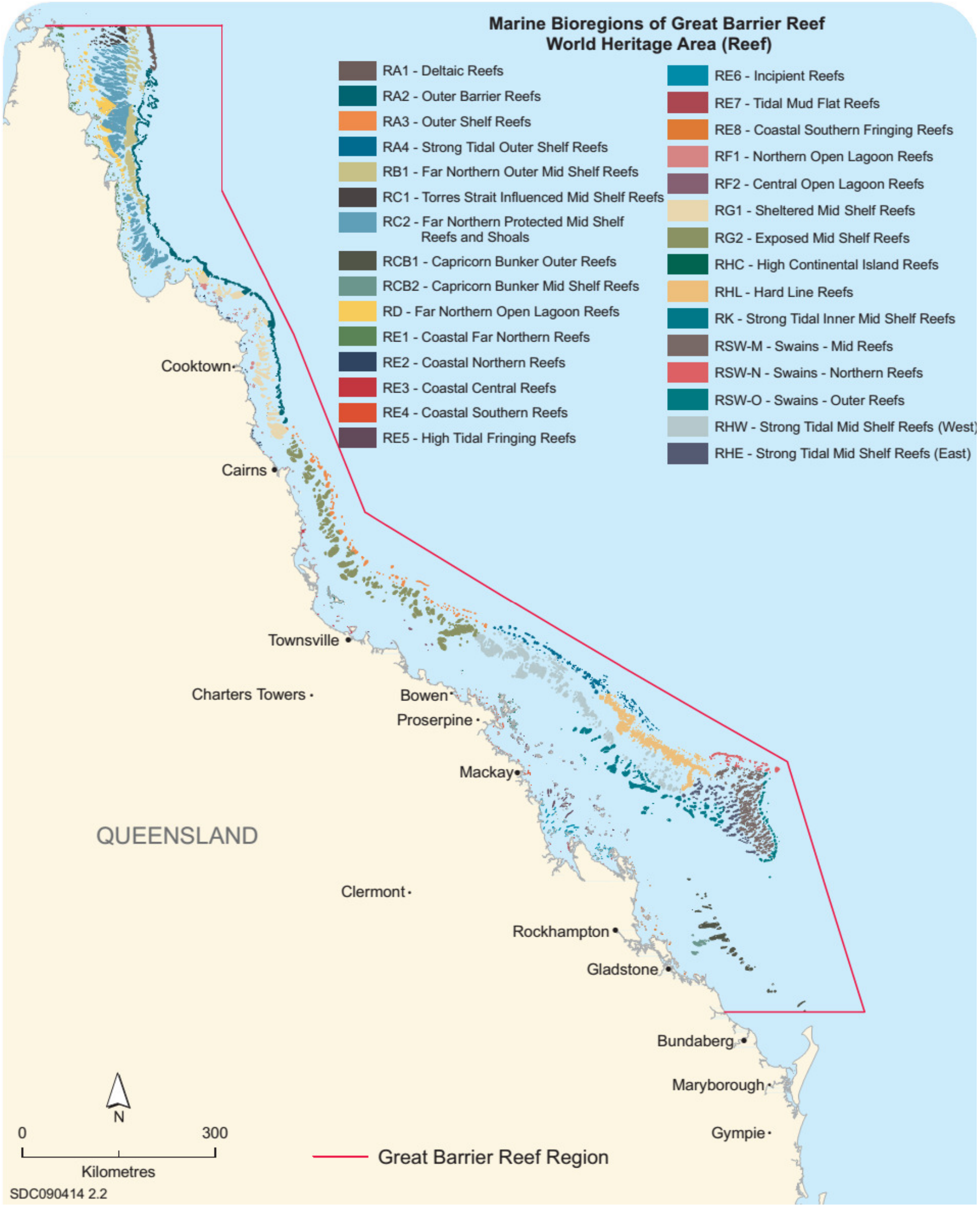


Figure 6: Reef Bioregions defined by GBRMPA expert taskforces as part of the preparation for the Representative Areas Program implemented in 2004. Source: GBRMPA (2009).

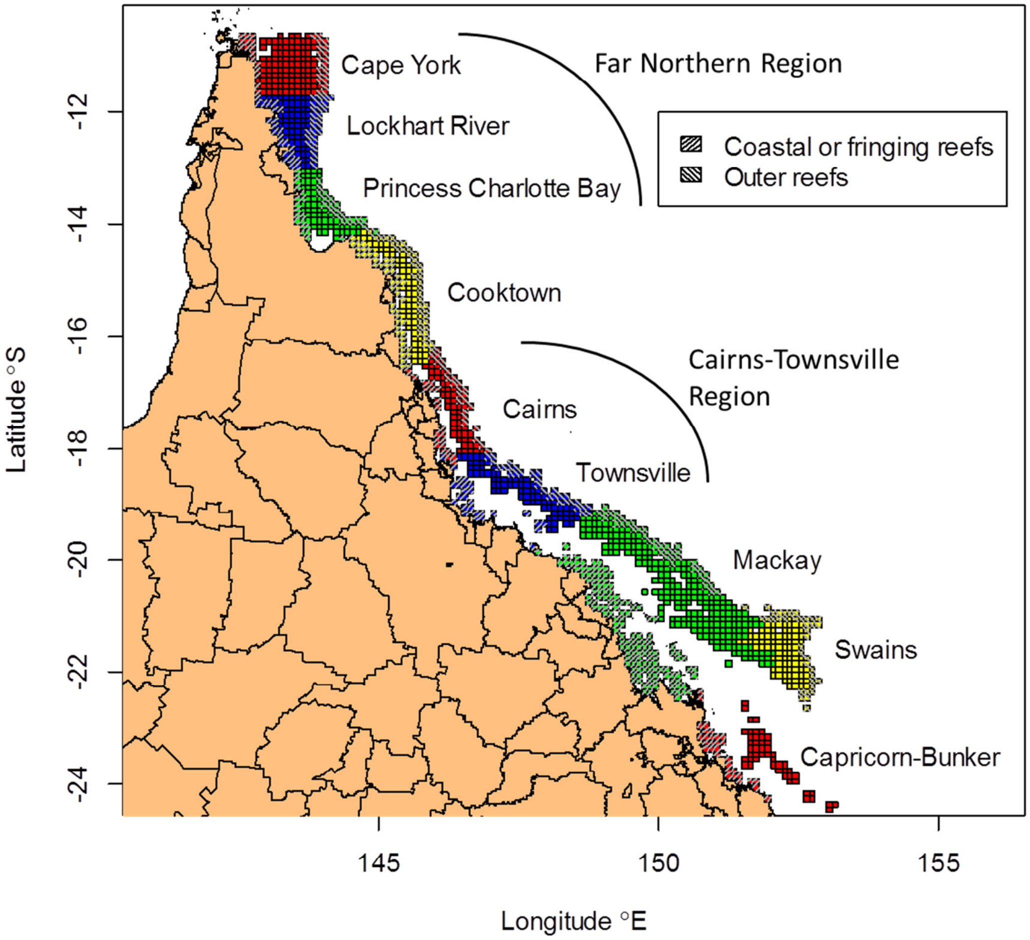


Figure 7: Map showing the Regions and Subregions used in the stock assessment. Regions are based on the Reef Bioregions in Figure 6. Because the fishing intensity increases from north to south in the northern regions, the Far Northern Region is divided into three Subregions, and the Cairns–Townsville Region is divided into two Subregions. The small squares on the map are the six-nautical-mile fishery logbook grid squares. Colours are chosen only to distinguish the Regions and Subregions, and have no other meaning.

1.3 The fishery

1.3.1 Description

The commercial Coral Reef Fin Fish Fishery targets mainly common coral trout (*Plectropomus leopardus*) for export live to Asia (see Figure 8). Secondary target fish include other species of coral trout (see section 1.3.2 below), red throat emperor (*Lethrinus miniatus*), and a large number of species of reef fish grouped together into the "Other Species" category for the purposes of management of the commercial fishery.

Around 2000 t p.a. of coral trout were landed by the commercial sector in the early 2000s (see Figure 9). Individual Transferable Quotas were introduced in 2004, at the same time as the GBRMPA Representative Areas Program, since when the annual Total Allowable Commercial Catch (TACC, the sum of all quota holdings that are available to be caught) has been about 1300 t. In calendar year 2011 only about half the TACC was caught, due to a combination of low catch rates, high fuel prices and low product prices. Both catch rates and product prices improved in 2013. The current commercial sector gross value of production (GVP) is roughly $30 million p.a., produced from about 250 licences.

Figure 8: The ultimate destination for most of the coral trout caught on the GBR: restaurants in Asia. This is the holding tank of a hotel restaurant in Shanghai. Source: Howard Norfolk www.aquarticles.com, licence similar to Creative Commons by Attribution for non-commercial use.

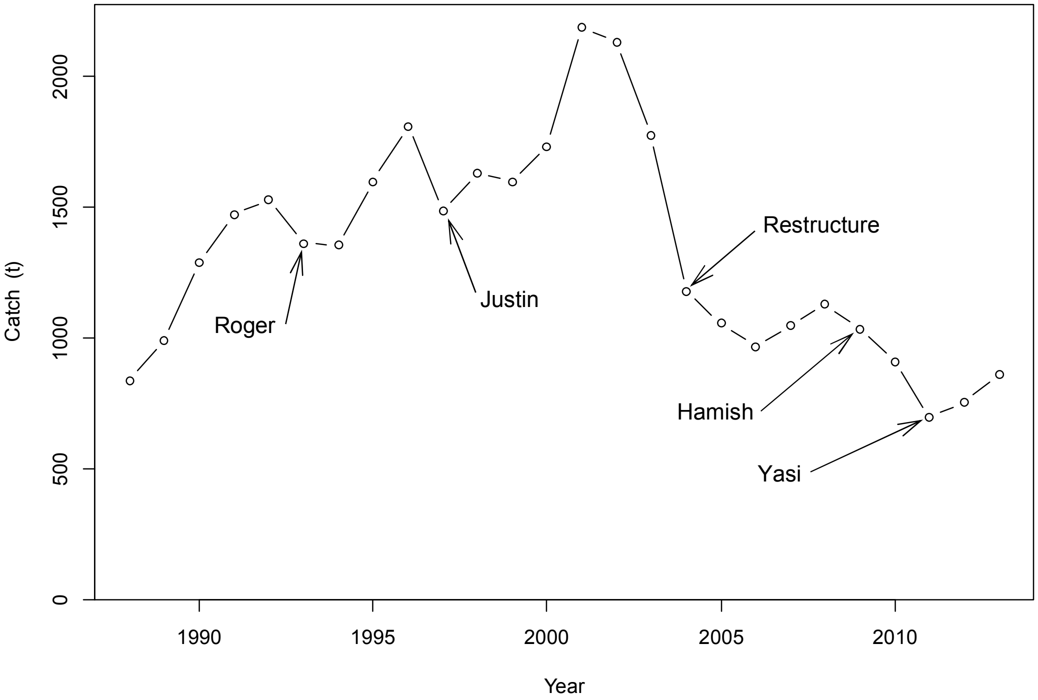


Figure 9: Annual commercial harvest of coral trout (several species combined) since the logbook system began in 1988. The 2004 fishery restructure and the major tropical cyclones to hit the fishery are also shown.

About 160 t p.a. of coral trout (several species combined) were landed by the recreational sector in 2011 (Taylor et al., 2012), down from about 500 t in 2000. The catch of common coral trout (*P. leopardus*) is roughly half of this total, with the rest mainly bar-cheek coral trout (*P. maculatus*). Indigenous catch of coral trout (all species combined) was about 7000 fish or 11 t in 2000 (Henry and Lyle, 2003, p. 121).

Most usage of the commercial quota (about 70%) is by lease-dependent fishers, who lease what they need annually from investors. The remaining usage is mainly by fishers who own roughly enough quota to cover the needs of their own fishing businesses (Thébaud et al., 2014, Fig. 6).

Concerns about overfishing of the Great Barrier Reef have been held since the Great Barrier Reef Marine Park was declared in the 1970s (Goeden, 1979). At the opposite end of the spectrum are views that the GBR is too big and robust for current levels of fishing or any other human activity to have any effect on it (e.g., Starck, 2005).

Tropical cyclones are believed to have a major effect on this fishery. One commercial fisher stated to us that Cyclone Hamish in 2009 loosened coral reefs, and then Anthony and Yasi in 2011 destroyed them. Other fishers said that recent cyclones had physically destroyed the fishery potential of some reefs, including Black's Reef [officially called Block Reef, reef number 19-127] and Oublier Reef [Outlier Reef, 19-120].

Many fishers believe that the commercial fishery is in economic decline, through a combination of cyclones, other weather events, perceived over-regulation, and possibly overfishing of accessible reefs. On fisher commented, "A scientific report said that any one of green zones, Individual Transferable Quotas and a limit on the number of licences would make the fishery biologically sustainable. We got all three of them, so it's no wonder that the fishery is struggling to make a profit." [We have not been able to locate the report referred to by this fisher.]

Lack of availability of skilled commercial fishers (dory operators) is also seen by fishers as a major limitation on the fishery. Many skilled dory operators have left the fishing industry in recent years to work in mining or farming.

On the current status of stocks, one commercial fisher said that fish weren't there like they used to be, but that abundance was increasing as the fishery recovered from recent cyclones. Another fisher stated that vessels based in Mackay had fished out the bigger coral trout close to home by 1984, then travelled to the Swain Reefs with hooks baited with pilchards. Before that time, specialist Swains fishers were able to catch all the fish they needed by wogging (working a specially-designed lure or "wog" towed behind a dory; see Williams and Russ, 1994, ch. 9).

The weather in 2012 was viewed by one fisher as similar to 1982–83, with strong southeast winds. It was generally recognised that coral trout tend not to bite in such weather.

There was a strong view among commercial fishers that fishery managers need to consider the level of total allowable commercial catch (TACC) in conjunction with *economic* sustainability of the fishery, not only biological sustainability.

In contrast to the state of the fishery, most fishers to whom we spoke while conducting this assessment believed that breeding stocks of coral trout on the GBR were healthy. They stated that coral trout was a prolific breeder; levels of recruitment were high; bommies [bomboras, characteristic of reef patches] down to 30m depth formed habitat for a huge breeding stocks; and that there was a huge population of coral trout in deep water, which was the major breeding stock. One fisher said that the 38cm minimum legal size for coral trout [increased from 35 cm in 1993] was the most positive thing that management had ever done. Commercial fishers generally believed that recreational fishing had no noticeable effect on commercial catch rates.

Fishers generally support the concept of green zones to protect the stock, but view those introduced by the Representative Areas Program in July 2004 as unnecessarily disruptive because they fragmented the available fishing grounds. Fishers prefer to have whole clusters of reefs zoned blue so that if fish won't bite on one reef the fishers can quickly move to a neighbouring reef.

Commercial fishers believe overwhelmingly that large amounts of fishing take place inside green zones. One fisher put the rate of noncompliance at about 80% of commercial fishers, and thought that between 20% and 50% of the entire commercial catch on the GBR came from green zones. Another comment was that the courts didn't impose effective penalties. A fisher may take a catch of $80k in a green zone, and pay fines of only $20k. A scientific tagging study also found that fishing in green zones was the only plausible way to explain the results (Davies, 2000).

Some fishers were also concerned about potential underreporting of catch taken by lease-dependent fishers, in order to avoid the costs of leasing extra quota. It was difficult to quantify the level of this underreporting.

The following comments by commercial fishers provide background to fishing techniques on the GBR:

- The strong tidal currents in the Mackay-Capricorn Section (from Mackay to the outer Swain Reefs; see section 1.2 and Figure 5) make these reefs more productive but also make fishing difficult. Few "hangs" (precise fishing locations) are suitable for fishing in strong currents. Most hangs require relatively small tidal runs.
- The Central Section (Innisfail to Mackay) is very productive for fishing, with plenty of bommies and submerged reefs. The most productive region of the GBR is around Bowen, especially the area comprising Darley Reef [19-043], Dingo Reef [19-038] and Tiger Reef [19-054]. These reefs could be fished on all tides and in high winds. [Tiger Reef was closed to fishing in the 2004 rezoning.]
- The Cairns Section (Lizard Island to Innisfail) and Far Northern Section (Cape York to Lizard Island) have smaller tidal ranges and are less productive.
- The Far Northern Section is qualitatively different from the other Sections. It has more sandy habitat and planar reefs. Catching coral trout in sandy habitat is a specialised skill.
- View buckets, which allow fishers to visually target fish, came into use on the Great Barrier Reef after 1990. Fishers based in Bowen have used them only since about 2002–2005, but fishers in Mackay have used them for longer.
- Chart plotters save about half an hour per hang, by allowing dory operators to find the right spot quickly.

1.3.2 Management history

Management of the fishery is the responsibility of Fisheries Queensland, an agency of the State of Queensland and a division of the Department of Agriculture, Fisheries and Forestry (DAFF Queensland). Management of the Great Barrier Reef Marine Park, on the other hand, is the responsibility of the Australian Federal Government, through the Great Barrier Reef Marine Park Authority (GBRMPA; see above). The two agencies collaborate on many aspects of fishery management within the Great Barrier Reef Marine Park.

The major fishery management measures implemented on the GBR are listed in Table 1. The fishery for coral trout developed later than other major reef fish species, and the first management measure specific to coral trout was introduced in 1976.

Zoning of the GBR, which involved declaring "pink zones" (no entry), "green zones" (closed to fishing), "yellow zones" (limited line fishing allowed), "blue zones" (line fishing allowed) and "sky-blue zones" (non-reef areas where trawling is allowed), was introduced Section by

Section between 1981 and 1988. About 5% of the GBR was closed to fishing before the 2004 rezoning, after which the proportion increased to about 33%.

Table 1: Significant events in the management of the Queensland Coral Reef Fin Fish Fishery and the zoning of the Great Barrier Reef Marine Park (GBRMP). Zoning Plans operated on Sections (Far Northern, Cairns, Townsville, Mackay–Capricorn, and Capricornia) which are unrelated to the later GBRMPA Bioregions on which the regional structure of this stock assessment is based. Major events not directly related to fishery management are also listed.

Date	**Management measures**	
		Other notable events
1877–1974	Various measures are implemented relating to fishing gear and practices.	
Approx. 1950		Commercial coral trout fishery begins.
1957	*Fisheries Act 1957* implements a minimum legal size (MLS) of 12 inches for red emperor (*Lutjanus sebae*) and "sweet lip" (red throat emperor, *Lethrinus miniatus*); the coral trout fishery is still in very early development and does not receive a MLS.	
1962		First records of commercial catches of coral trout are collected by the Queensland Fish Board (QFB).
16 Dec. 1976	*Fisheries Act 1976* implements MLS of 35 cm for coral trout (nominally *Plectropomus maculatus*, but species on the GBR were not well determined at that time), 35 cm for red emperor, 30 cm for red throat emperor.	
1 Jul. 1981		Zoning of Capricornia Section of GBRMP (Capricorn–Bunker reefs)
1 Jul. 1983	Replenishment closures of two Capricorn–Bunker reefs, North Reef (23-045A) and Boult Reef (23-079)	
7 Nov. 1983		Zoning of Cairns Section of GBRMP (Lizard Island to Innisfail)
1983–1986		Hundreds of reefs are surveyed by underwater visual surveys (UVS) under contract to GBRMPA to directly measure abundance of fish. Many surveys of smaller numbers of reefs are conducted on a one-off basis in subsequent years.
1 Feb. 1986		Zoning of Far Northern Section of GBRMP (Cape York to Lizard Island)
1 Jul. 1986	North Reef reopens.	
1 Dec. 1986	Boult Reef reopens.	
1 Oct. 1987		Zoning of Central Section of GBRMP (Innisfail to Mackay)
1 Jan. 1988	Commercial logbook database begins.	
1 Aug. 1988		Zoning of Mackay–Capricorn Section of GBRMP (Mackay to the Swains)
22 May 1990	Recreational fishers are prohibited from selling any of their catch (Higgs, 1995).	
Early 1990s		Export of live coral trout to Asia begins.
1 Jan. 1992	Replenishment closure of Bramble Reef (18-029) near Townsville	
3 Apr. 1992		Second zoning of Cairns Section of GBRMP, the only rezoning to open a significant number of reefs previously closed to fishing
1992		Regular UVS by the Australian Institute of Marine Science's Long Term Monitoring Program (AIMS LTMP) begins.
25 Jun. 1993	*Fishing Industry Organisation and Marketing Amendment Regulation* implements MLS of 38 cm for coral trout, 45 cm for red emperor, 35 cm for red throat emperor, first-time MLSs for many other reef fish species; recreational bag limits for many reef fish species, including 10 coral trout (*Plectropomus* spp., all species combined), 10 fish of any one species of common *Lethrinus* and *Lutjanus* species; bag limit 30 reef fish in total (all reef fish species combined) (allowed numbers were doubled for "extended fishing charters").	
1 Jul. 1995	Bramble Reef reopens.	
Mar. 1997		Tropical Cyclone Justin has a big effect on the fishery.

19 May 1997	Investment Warning is issued.	
1995–2005		Effects of Line Fishing (ELF) experiment studies four clusters each of six reefs, some of which have their zoning manipulated for experimental purposes; and undertakes the only scientific age-structure sampling of fish ever allowed in green zones.
15 Apr. 2002		Second zoning of Far Northern Section of GBRMP
12 Sep. 2003	*Fisheries (Coral Reef Fin Fish) Management Plan 2003* revises MLS of *Plectropomus laevis* to 50 cm with a maximum legal size of 80 cm, MLS of red throat emperor to 38 cm, MLS of red emperor to 55 cm; recreational bag limits to 7 coral trout (all coral trout species combined), 8 red throat emperor, 5 of any single emperor (Lethrinidae) species other than red throat, 5 red emperor, and 20 reef fish (all reef fish species combined) (higher numbers are allowed for fishing charters of extended duration); revises MLSs and bag limits for many other reef fish species.	
1 Jul. 2004		Representative Area Protection (RAP) and comprehensive rezoning of whole GBR; proportion of GBR closed to fishing increases from about 5% to 33%.
1 Jul. 2004	*Fisheries (Coral Reef Fin Fish) Management Plan 2003* introduces Total Allowable Commercial Catch (TACC) 1350 t (later reduced to 1288 t by buy-outs under the RAP) and Individual Transferable Quotas (ITQ)	
9 Oct. 2004	First eight-day coral-trout spawning closure begins under the *Fisheries (Coral Reef Fin Fish) Management Plan 2003*, which implements fishery closures around the new moon in October, November and December each year 2004–2008. From 2009 onwards these were reduced to five-day closures in October and November only.	
Mar. 2009		Severe Tropical Cyclone Hamish has a big effect on the fishery.
Feb. 2011		Severe Tropical Cyclone Yasi, one of the largest and most powerful in Queensland's history, strikes north Queensland.

1.4 Taxonomy of coral trout

Modern taxonomy of coral trout is based on the monograph of Randall and Hoese (1986), but is still "in a state of flux" according to Craig et al. (2011, p. xii).

Coral trout belongs to the category of fish called groupers, which make up the subfamily Epinephelinae within the family Epinephelidae. Until the early 2000s, the family now called Epinephelidae was classified as a subfamily (called Epinephelinae) of Serranidae, and groupers were classified as the tribe Epinephelini. Grouper specialists now recognise the family Epinephelidae, but many sources still place groupers in the family Serranidae.

Species of coral trout that may be present on the Great Barrier Reef (GBR) are listed in Table 2. Most of the fishery catch comprises species from the genus *Plectropomus*. Photographs of the major ones are shown in Figure 10. Coral trout change their colour according to their environment, and the species are difficult for non-experts to distinguish.

Plectropomus leopardus, *P. maculatus* and *P. laevis* are common on the GBR. Commercial fishers state that *P. areolatus* can make up a substantial part of catches in the Far Northern Section. Whether *P. oligacanthus* exists on the GBR is uncertain; for the purposes of stock assessment, this species can be ignored. *Variola louti* and *V. albimarginata* are present on the GBR, but are caught in much lower numbers than the three major *Plectropomus* species.

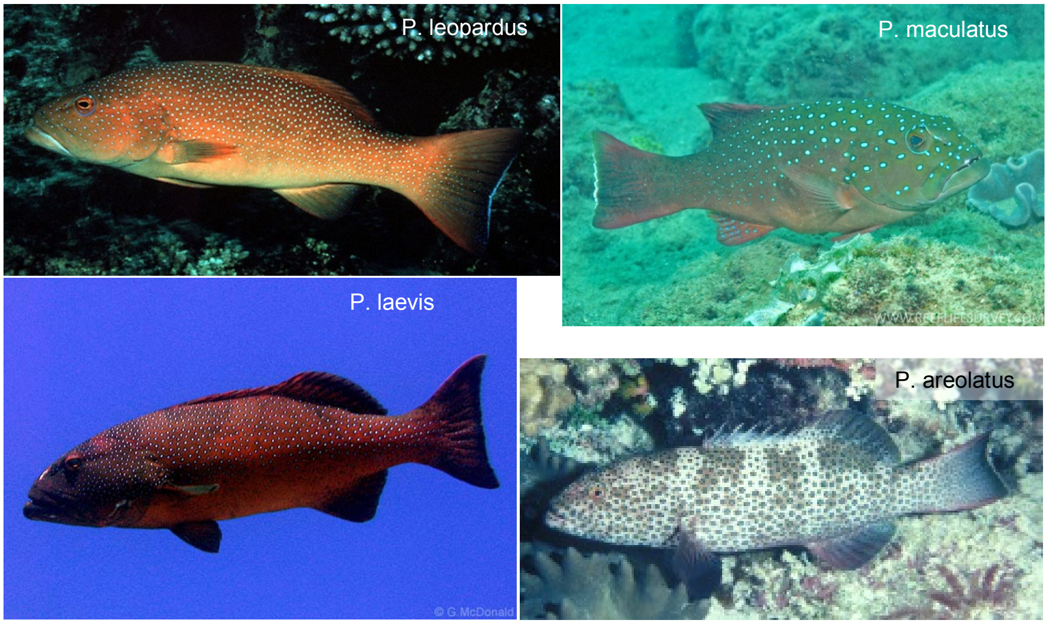


Figure 10: Major Plectropomus species present on the Great Barrier Reef. Sources: *P. leopardus* and *P. areolatus* Randall (1997); *P. maculatus* Rick Stuart-Smith, www.reeflifesurvey.com (Creative Commons by Attribution licence for non-commercial use); *P. laevis* G. McDonald, Cook Islands Biodiversity and Natural Heritage Database, http://cookislands.bishopmuseum.org (licence similar to Creative Commons by Attribution for non-commercial use).

Table 2: Species of coral trout that may be present on the Great Barrier Reef.

Scientific name	**Authority**	**Code**	**Standard name**
Plectropomus leopardus	Lacépède, 1802	37 311078	Common coral trout
Plectropomus maculatus	Bloch, 1790	37 311012	Bar-cheek coral trout
Plectropomus laevis	Lacépède, 1801	37 311079	Blue-spotted coral trout
Plectropomus areolatus	Rüppell, 1830	37 311081	Passionfruit coral trout
Plectropomus oligacanthus	Bleeker, 1854	37 311162	Vermicular cod
Variola louti	Forsskål, 1775	37 311166	Yellow-edge coronation trout
Variola albimarginata	Baissac, 1953	37 311026	White-edge coronation trout

1.5 Biology of coral trout

1.5.1 Biology not related to ageing

Coral trout on the GBR has been extensively studied over several decades. Early work was conducted in the 1970s (e.g., Goeden, 1978) but was limited by lack of accurate ageing techniques.

Coral trout is a high-level predator on the GBR and eats mostly fish from a wide variety of prey families, plus a small proportion of invertebrates (Goeden, 1978; Kingsford, 1992; St John et al., 2001). Common coral trout *Plectropomus leopardus* fetches a premium price because it is red, a lucky colour in Asia. Even this species, however, is coloured green when it occurs in lagoons, and commercial fishers don't target it there. Blue-spotted coral trout *P. laevis* when young has a different colour form (the "Chinese footballer" coloration of yellow, white and black), which does not appeal to Asian consumers. It also grows much larger than the preferred "dinner plate" size.

Predators including coral trout have a major effect on the forage fish (smaller fish) around coral reefs. Madin et al. (2011) present a particularly graphic example from the GBR showing "reef halos" around coral reefs where forage fish have eaten the algae. Further from the reefs (beyond about 30 m), the algae are dense because forage fish have nowhere to hide from predators and do not venture that far from safe territory. The authors present another example from a very heavily fished reef in Indonesia where no halos are apparent: forage fish venture far from reefs because the predator population has been greatly depleted.

Evans et al. (2010) conducted a genetic study of coral trout on the GBR and found no spatial separation of stocks. There is, however, only a small range of movement of adults and larvae (see below). Full spatial mixing over the GBR is possible but would take many generations.

Coral trout species are protogynous hermaphrodites, beginning as female and later changing sex to male (although some fish may become male prior to maturity) (Loubens, 1980; Russ, 1984; Ferreira, 1993; 1995; Brown et al., 1994). It is a matter of continuing debate whether, on the one hand, the sex ratio is controlled socially to compensate for potential removal of larger, predominantly male fish or, on the other hand, preferential extraction of these fish by fishing reduces the proportion of males. Coleman et al. (1996) found that, for groupers in the Gulf of Mexico, persistent fishing of spawning aggregations greatly reduced the proportion of male gag and scamp groupers (*Mycteroperca microlepis* and *M. phenax*), but fishing had had no significant effect on the sex ratio of red grouper (*Epinephelus morio*) which does not aggregate. Mackie (2003) found that the sex ratio of half-moon grouper (*Epinephelus rivulatus*) on Ningaloo Reef in Western Australia was socially controlled.

On the Great Barrier Reef, Adams et al. (2000) found no statistically significant difference in sex ratio of *Plectropomus leopardus* between green zones and blue zones. This finding indicates that sex ratio may be socially controlled, although alternative explanations are that there may be extensive fishing in green zones, and that the level of fishing even in blue zones of the GBR may not be high enough to significantly alter the sex ratio.

Scientific studies indicate that coral trout spawn in the spring and summer. Brown et al. (1994) state, "Coral trout spawn during the period from early spring to summer—September–December in the northern part of the GBR and October–February in the south." Samoilys (2000) concurred with the September–December spawning period in the northern GBR, and noted that spawning aggregations generally coincided with the new moon.

Commercial fishers state that coral trout tend not to take bait after August in the southern GBR and that fishers have to gradually move north late in the year to catch coral trout (commercial reef fishers, personal communications, 2012). Perhaps there is not a strong correlation between spawning and willingness to take bait.

Coral trout have been known to move between reefs after they have settled as juveniles, but infrequently. Davies (2000) found that only one of 128 research returns of tagged *Plectropomus leopardus* had moved to a reef different from the reef of release, and this was to an adjacent reef separated by a channel only 200 m wide. The apparent rate of migration in tag returns from the public was much higher, and was thought to be inaccurate. There was, however, an indication that *P. leopardus* may move between reefs to and from the location of a spawning aggregation.

Other tagging studies have concentrated on movement within the same reef on which fish were tagged. Zeller et al. (2003) found that only 13% of tagged fish recaptured within two years were more than 50 m from the point of tagging. Samoilys (1997) found movement up to 7.5 km, still within the reef of release.

Larval movement of coral trout is somewhat greater. Harrison et al. (2012) conducted a genetic parentage analysis of potential bar-cheek coral trout *P. maculatus* parents in green zones and offspring in green and blue zones within 30 km. They found that about 30% of juveniles with assigned parents were collected within 2 km of the parents; one juvenile was 28 km from the parents, and the average was 8.6 km. We note that this study did not include a scientific control of potential parents from blue zones as a check on the methodology, so some researchers may question the results.

Previously it was hypothesised that reef-fish larvae could be transported long distances on prevailing currents (Williams et al., 1984) or by swimming (Stobutzki and Bellwood, 1997; Fisher, 2005). The swimming ability of larvae is certainly sufficient to aid settlement on a reef in preference to a non-reef location (Atema et al., 2002), but the larvae may in fact use it to stay on or close to their natal reefs and *avoid* being transported elsewhere by currents (Gerlach et al., 2007).

Post-release survival of reef fish, including coral trout, after the fish had been caught by hook and line, was studied by Brown et al. (2008). Chapter 2 of their report describes an experiment in which fish were kept in enclosures for three days, while Chapter 4 compares recapture rates of tagged animals and has appeared as a journal article (Sumpton et al., 2010).

For *P. leopardus*, Brown et al. found that capture depth and fish length were the most important factors affecting survival. Survival rates from the enclosure experiment, adjusted for various other factors, were 91% for fish initially caught at depth 1 to 19 metres, and 41% for fish initially caught at depth 20 m or greater. By fish length, the survival rates were 83% for legal-sized fish (≥ 38 cm total length, ≥ 36 cm fork length), and 66% for undersized fish. It was hypothesised that the lower rate for undersized fish occurs because smaller fish don’t fight as hard, are brought to the surface more quickly, and hence are more likely to suffer barotrauma (trauma from decompression). Results from tagging were in accord with these results, although the absolute rates of survival could not be estimated from the tagging experiment because they were confounded with the reporting rate of tag recoveries by fishers.

For use in stock assessment of a fishery that is primarily commercial, the survival rate of released fish should be that of undersized fish caught in shallow water. The assumptions here are that the commercial fishery will discard only undersized fish, and that because it targets the live-fish trade it will aim to catch fish whose survival will not be compromised by capture depth. Then the results above indicate a survival rate of roughly 75%.

1.5.2 Biology related to ageing

Accurate ageing techniques based on annual rings in otoliths (“ear bones”) were developed for tropical reef fish in the late 1970s (Loubens, 1978). The first major studies to employ them on the GBR took place in the 1980s and early 1990s (McPherson et al., 1985; Brown et al., 1994; Russ et al., 1995). These studies developed a preference for otoliths that had been sectioned by sawing, especially for older fish and especially for coral trout, whose otoliths are less translucent than those of other reef fish such as Lethrinids (emperors) (Mapstone et al., 2004b).

Annual recruitment strength of coral trout can vary greatly. Russ et al. (1995; 1996; 1998) describe an extremely strong cohort (year-class) of *P. leopardus* that was spawned in late 1983, settled on the reefs in early 1984, and dominated the population off Townsville for some years.

The data that identify the above strong cohort act as a validation of the ageing technique used by Russ et al. (1995). In this technique a ring (which is opaque) is counted only when it has a translucent outer margin; rings formed in the year of sampling are not counted. Russ et al. conducted their sampling between June and October from 1990 to 1992. The 1993 sampling was actually conducted between March and May 1994 and the newly-laid 1993 ring was not counted.

Subsequent experiments, notably the ELF Project and the Fisheries Queensland sampling (see Table 1 and section 1.8), collected fish between October and December each year. The ELF Project used the same ageing protocol as Russ et al. (1995) (see Mapstone et al., 2004b). Because the fish were collected later in the year than the Russ et al. samples, some newly-formed rings may have been counted. This matter is not pursued further in this report, and we assume that the current year's ring is not counted, but it could be investigated in future.

The Fisheries Queensland otolith reading protocol recorded the otolith edge type (narrow, intermediate or wide translucent margin) in addition to the number of rings. Narrow edges were counted as newly formed in the year of sampling; intermediate edges were counted as formed in the previous year if sampled before November but newly formed in November and December; and wide edges were counted as formed in the previous year if sampled before December but newly formed if sampled in December.

Ferreira and Russ (1992) derived a von Bertalanffy growth curve for *P. maculatus* from inshore reefs of Townsville, with parameters $L_\infty = 60.0$ cm standard length, $K = 0.206\ \mathrm{yr}^{-1}$ and $t_0 = -0.945$ yr. The von Bertalanffy growth curve (von Bertalanffy, 1938) takes the form

$$L = L_\infty \left[1 - \exp\{-K(t - t_0)\}\right],$$

where L_∞ is the theoretical maximum length, K is the growth-rate parameter, and t_0 is the theoretical age at which a fish has length zero. In practice, the curve is fitted to a data set that includes neither tiny fish nor fish of age infinity: it is simply a fit to available data, and the parameters should not be interpreted literally. Also the maximum length to which a fish will grow varies substantially between individual fish, so it is common to observe lengths greater than L_∞.

The oldest specimen of *P. maculatus* examined by Ferreira and Russ, which was also the largest, was a male aged 12 years with a standard length of 58 cm. Fork length (FL, length measured from the front tip of a fish's head to the fork of its tail), which is the measure more commonly used in recent years, is somewhat greater than standard length (SL, measured to the end of the final vertebra), and they fitted the following relationship between the two: $\mathrm{SL} = -0.4236 + 0.8565\,\mathrm{FL}$, both measured in cm.

For *P. leopardus*, Ferreira and Russ (1994) derived a von Bertalanffy growth curve from samples from Lizard Island in the northern GBR, with parameters $L_\infty = 52.2$ cm fork length, $K = 0.354\ \mathrm{yr}^{-1}$ and $t_0 = -0.766$ yr. There was a large amount of variation about the mean length at age (ranges of about ±10 cm, standard deviation of roughly 5 cm), implying a fairly poor relationship between length and age. The oldest fish observed was 14 years of age; it was also the equal largest with a fork length of about 62 cm, which was matched by another fish aged 9 years. The relationship between standard length and fork length (both in cm) was $\mathrm{SL} = -0.308 + 0.852\,\mathrm{FL}$, and that between total weight (TW, in kg) and fork length (in cm) was $\mathrm{TW} = 7.9 \times 10^{-6}\,\mathrm{FL}^{3.157}$.

Heupel et al. (2010) derived growth parameters for *P. laevis* sampled from the whole range of the GBR: $L_\infty = 115.9$ cm fork length, $K = 0.096\ \mathrm{year}^{-1}$, $t_0 = -2.28$ yr. The oldest fish observed was 15 years. The only available fish for this analysis were taken as by-product of line-fishing experiments targeting the smaller species *P. leopardus*; therefore the analysed sample lacked both the biggest and smallest fish in the population, due to (a) the fishing gear used, and (b) the lack of an auxiliary targeted sample of very small *P. laevis* as was undertaken for the above studies of *P. maculatus* and *P. laevis*. The relationship between total weight (TW, in kg) and fork length (in cm) was $\mathrm{TW} = 3.8 \times 10^{-9}\,(10 \times \mathrm{FL})^{3.21}$.

Sex ratio of *P. leopardus* was examined by Adams et al. (2000), Ferreira (1995) and Russ et al. (1995). Adams et al. (2000) found large differences in sex ratio between individual reefs, with measurements ranging from 0.3 to 5.5 females per male, and they found a higher proportion of males in the Swains region than on reefs off Townsville. The ages of fish transitioning from female to male were mostly 4–6 years in the Swains and 3–5 off Townsville, but a few transitional fish in the Swains were 7–9 years old. Females were observed at ages 2–10 years in the Swains and 1–9 off Townsville; for males the ranges were 2–17 and 2–10 respectively.

Ferreira (1995) found sex ratios of about 2.5 in reefs off Townsville and in reefs adjacent to Lizard Island, but do not present results for individual reefs. Transitional fish were observed at ages 2–6 years off Lizard Island, and 3–11 off Townsville. Females were observed at ages 2–11 off Lizard Island and 2–10 off Townsville, although most females were aged 2–7 and 2–6 respectively. Males were observed at ages 2–14 in both regions.

Russ et al. (1995) found sex ratios ranging from 1.4 to 2.8 in reefs off Townsville. Transitional fish were observed at ages 2–14 years. Mature females were observed at ages 2–13, mostly in the range 4–9. Mature males were observed at ages 2–14, mostly in the range 6–10.

The samples collected by Russ et al. (1995) consist predominantly of older fish. The obvious explanation is that the strong 1984 cohort has supressed the strength of subsequent cohorts, through either competition or cannibalism. The younger age classes that are present show a much higher sex ratio (fewer male fish) compared to the same age classes in the ELF Project data set (described in Table 1 and section 1.8). This observation strongly supports the hypothesis that sex ratio is socially controlled in *P. leopardus*. The high numbers of older fish have already produced plenty of males, so younger fish remain female.

Coral trout mature early in life. Ferreira (1995) found that at reefs off Lizard Island about 80% of female *P. leopardus* aged 2 and 3 years were mature, and all females aged 4 or more were mature. Off Townsville the figures were about 25% mature at age 2, 60% at age 3, 90% at age 4 and 100% at age 5 or more. Russ et al. (1995) found roughly 20% maturity at age 2, 60% at age 3, 90% at age 4, 98% at age 5 and 100% at age 6 or more.

Ferreira (1993) found a sex ratio of 2.6 for *P. maculatus* in reefs off Townsville. The observed transitional fish were all aged 3 years; females were observed in the age range 1–7, and males in the range 3–12.

For *P. laevis*, Heupel et al. (2010) observed transitional fish from ages 1–4 years, female fish 1–13 and male fish 2–14, although most females were aged 3–6 and most males 6–10.

Fecundity (number of eggs produced) of *P. leopardus* was examined by Carter et al. (2009a) over the GBR, using samples from the ELF Project. They found a stronger relationship of fecundity with weight than with age, indicating that fecundity may be primarily size-dependent rather than age-dependent. The relationship with weight showed that fecundity increased faster than weight as the fish grew: fecundity was proportional to weight to the power 1.56 (Carter et al., 2009a, Fig. 1(b)).

Carter et al. (2009a) claim that their methodology is superior to earlier methodology used by Samoilys (2000) (see Carter et al., 2009b). Samoilys (2000) analysed a much smaller sample from two reefs off Cairns and found a relationship of fecundity proportional to weight to the power 0.7946. We do not regard an exponent substantially less than 1 as credible because it is well recognised in fish biology that relative fecundity increases with age and size.

The fecundity results of both Carter et al. (2009a) and Samoilys (2000) are based on quite weak regressions with large amounts of random variation about the fitted relationships (R^2 = 0.22 and 0.37 respectively). Possibly this is because for coral trout it is difficult to exactly synchronise the sample collection with the spawning time. Also the ELF Project sampling was deliberately timed to coincide with the full moon, not the new moon, in order to avoid spawning aggregations (Mapstone et al., 2004b).

1.5.3 Modelling decisions and new growth parameters

Published relationships on fish growth of *P. leopardus* were found not to fit well to available data on length and age collected by the ELF Project and Fisheries Queensland (data sets are described in detail in the next section). Therefore new relationships were derived from the ELF data. The ELF Project attempted to use the same fishing methods and equipment as the commercial coral trout fishery, and so was regarded as the best data set for this purpose. It caught larger coral trout on average than the Fisheries Queensland sampling. The following relationships were fitted to the ELF data:

- Von Bertalanffy growth curve parameters: $L_\infty = 66.33\,\text{cm}$, $K = 0.1005\,\text{yr}^{-1}$, $t_0 = -5.256\,\text{yr}$. If the current year's ring is counted, increasing all the ages by one year, then one year also has to be added to t_0, making $t_0 = -4.256\,\text{yr}$.
- Coefficient of variation (ratio of standard deviation to mean) of individual fish length about the above growth curve: 0.07 at age 1, rising linearly with age to 0.11 at age 19.
- Length–weight relationship (log-linear regression): $\text{TW} = 6.850\times10^{-06}\ \text{FL}^{3.1964}$.
- Coefficient of variation (standard deviation parameter of the log-linear regression) of individual fish weight about the above length–weight relationship: 0.143.
- Average weight of a legal-sized *P. leopardus*: 1.5760 kg.

Mapstone et al. (2004a) published data on 24 fish of length close to the MLS, from which we fitted the following relationship between fork length and total length (log-linear regression):

- Fork-length–total-length relationship: FL = 0.9409 TL.
- Coefficient of variation of individual fish length about the regression: 0.0042.

This regression is needed because the MLS is for total length, and the fish length estimated by divers conducting underwater visual surveys is total length, whereas most scientific analysis uses fork length.

Other decisions on the biology of *P. leopardus* for population modelling purposes were taken as follows:

- Fish do not move from the reef on which they settle as juveniles. For modelling purposes this means that they remain in the same Subbioregion and they do not move between blue zones and green zones.
- Adult spawners in green zones contribute to recruitment in blue zones.
- Sex ratio is socially controlled. Therefore sexes can be combined, and sex ratio as a function of age does not need to be included in the model.
- Fecundity (relative contribution of a fish to spawning) is proportional to fish weight raised to the power 1.56, as in Carter et al. (2009a), up to age 13 years (14 if the current year's ring is counted), which was the age of the oldest female fish observed by Russ et al. (1995); fecundity is constant thereafter (all ages ≥ 13 or 14).
- The proportion of fish that were mature, as a function of age, was fixed at a compromise of the two regions sampled by Ferreira (1995), one of which was also sampled by Russ et al. (1995): zero at ages 0 and 1, 40% at age 2, 70% at age 3, 95% at age 4, 99% at age 5, and 100% at age 6 and above.
- The survival rate of undersized fish released by fishers was fixed at 75%.

All biological relationships were assumed to apply to *P. leopardus* over the whole GBR; no region-specific relationships were used. The data suggested that population parameters on reefs off Townsville may be different to those elsewhere on the GBR, but we were unwilling to postulate relationships that did not change smoothly with latitude and, moreover, may not even apply in other parts of the Cairns–Townsville Region.

We emphasise that the relationships derived above, especially the growth curve, are applicable only to the fish sampled, which were caught by hook and line. They are not applicable to very young or very small fish which will not take bait on a hook of the size used in the commercial fishery.

The oldest and longest individual fish caught in the age-frequency data sets were

- Russ et al. (1995), 14 years, 60–64 cm FL interval
- ELF Project, 18 years (19 if current year's ring is counted), 66.2 cm FL
- Fisheries Queensland, 18 years, 65 cm FL.

One record thought to be a data error was excluded (age 30 years, fork length 34.7 cm).

1.6 Social learning in fish populations

Social learning is the process whereby fish are able to learn from each other to acquire skills and knowledge to become better able to succeed in life. Brown and Laland (2011) define it as "any incidence in which individuals acquire new behaviour or information about their environment via observation of, or interaction with, other animals or their products." Study of social learning in fish populations has increased enormously over the last few decades (Brown et al., 2011a, ch. 1; Laland et al., 2011).

Ideas about the intelligence of fish populations that 30 years ago would have been regarded as fantasy are now well accepted. Brown et al. (2011b, p. 74) state, "Carefully controlled laboratory studies have shown an exceptionally high degree of sophistication in the learning abilities of prey fishes. … The prey learns to recognise not only the predator as a threat, but also the level of threat posed by the predator, making it possible for the prey to match the intensity of their antipredator response to the risk posed by the predator. Prey fishes continually update information regarding the risk level of predators and learn the temporal foraging patterns of their predators. Prey fishes can generalise recognition of one predator to other similar species and hence avoid some of the costs associated with learning." Acquired knowledge for defence against predators is thought to be retained on a scale of weeks to months (Brown et al., 2011b, p. 70; Nilsson et al., 2008), but sometimes years (Magurran, 1990).

As implied above, much of the literature on defence mechanisms of fish populations concerns predator avoidance (Brown et al., 2011a, ch. 3, 4, 11). Much less work seems to have been done on defence against human activities such as fishing, although the difference between that and predator avoidance may seem small. Channels of communication such as chemosensory cues that warn against predators could be expected to do the same against dangers posed by fishing.

Some studies have been conducted on avoidance of divers by reef fish, including groupers, on reefs that are spear-fished (Januchowski-Hartley et al., 2011), and have found that the flight initiation distance (distance from a diver within which fish will flee) increases with the level of fishing. Researchers have ascribed the flight response to social learning that divers constitute a danger, although this is not yet proven; an alternative explanation is that individual fish simply flee a large object, dangerous or not, earlier when they have seen it before.

Hook shyness has been studied for freshwater trout (which is unrelated to coral trout, despite the similarity of name) (Young and Hayes, 2004; Askey et al., 2006). These studies concentrate on individual learning rather than social learning: the individuals that become hook-shy (difficult to catch as a result of human fishing activity) are assumed to be individuals that have previously been caught and released.

Social learning, on the other hand, applies to individuals that have not had a direct experience such as being caught and released, but have acquired knowledge of the risk from their fellows. The potential for social learning in freshwater trout populations does not seem to have been seriously studied.

Other studies on social learning in groupers have focussed on how groupers learn the routes to spawning sites (Bolden, 2000). Capabilities other than travel to spawning sites and avoidance of potential spear fishers have to be inferred from other fish families.

Commercial fishers on the GBR believe that coral trout are capable of social learning and that, when a population is fished, individuals quickly learn not to take bait. As one fisher put it, "When they handed out brains to fish, coral trout got their fair share." Other comments were as follows:

- Fishers need to keep moving, not go back to the same spot every day. They also need to move on to the next bommie if the fish on one bommie won't bite. Many fishers stay too long trying to catch fish that they can see through their view buckets.
- It is rare to get two good days in a row in one area. A yield of 150 fish per day quickly falls to 15 per day, because the fish stop biting.
- There are some spots where coral trout won't bite on a hook. Too many boats travelling around make them hook shy.
- Catch rates are higher in the Far Northern Section of the Great Barrier Reef because there are fewer boats. Fish are not being spooked by the presence of too many boats.
- Fish are still naïve (uneducated) in the Far Northern Section. A good fisher can catch 50 coral trout in a day on a single bommie. Fish are smarter around Bowen, because the population has been fished intensively.
- The fishery could be made more profitable by managing it to take only the "cream" off each reef [the fish that are easy to catch]. There will be plenty of fish remaining, but they will be difficult and expensive to catch.
- Green zones [zones closed to fishing] have had a negative effect on the fishery because fishers now have to fish the blue zones [zones open to fishing] more intensively, and the fish there have learnt not to take bait.
- Fish may move from blue zones into green zones because they know that they will be safer in green zones.

1.7 Other grouper stock assessments

Quantitative stock assessments of other grouper fisheries have been carried out principally in North America:

- Gag grouper *Mycteroperca microlepis* (NOAA, 2006b)
- Red grouper *Epinephelus morio* (NOAA, 2006a)
- Goliath grouper *Epinephelus itajara* (NOAA, 2011).

Groupers in North America have a somewhat different life cycle to those in Australia. They tend to live in shallow water as juveniles and then move offshore into deep water as they mature (roughly 30–160 m depth for gag grouper; 50–300 m for red grouper; 1–100 m for goliath grouper). They generally grow larger than coral trout, with estimated L_∞ parameters of 131 cm (gag grouper), 85 cm (red grouper) and 238 cm (goliath grouper). Gag grouper matures at about age 3 years, red grouper about 3–6 years, and goliath grouper about 5–7 years). Maximum ages observed appear to be 31 years for gag grouper, 24 for red grouper and 37 for goliath grouper.

Goliath grouper has suffered severed population declines from overfishing. The maximum lifespan is thought to be considerably greater than the 37 years observed.

For the purposes of this coral trout stock assessment, the important parameter from the North American assessments is the recruitment compensation ratio, denoted r (Goodyear, 1977), which is the average number of offspring of each adult fish that survive to spawning age, when the population size is very low. It is a measure of the productivity of the population when population size is not limited by competition between individuals.

It is always the case the $r > 1$ because if the population is ever going to recover, the average adult must do more than simply replace itself. An equivalent parameter known as steepness, denoted h, came into use later than r and is defined as $h = r/(4+r)$; it lies in the range $0.2 < h \leq 1$, and is the ratio of recruitment to virgin recruitment when stock size is reduced to 20% of the virgin size. The limiting case $r = \infty$ ($h = 1$) corresponds to recruitment that does not depend on the spawning stock size, so that the population is infinitely productive in the sense that fishing down the number of spawners has no effect on recruitment.

If all other things are equal, a higher value of r means that the population can withstand a higher level of fishing. When comparing different species, allowances must be made for the age at maturity and the longevity of the fish. Long-lived fish and fish that mature later in life tend to be more susceptible to overfishing.

The recruitment compensation ratio is generally difficult to estimate in a stock assessment, and assessments often assume particular values for it. The values used in the North American stock assessments were the following:

- Gag grouper: base value $r = 12.0$, smallest value tested $r = 7.9$.
- Red grouper: base value $r = 25.2$, smallest value tested $r = 6.0$.
- Goliath grouper: base value $r = 22.0$, smallest value tested $r = 8.8$.

Coral trout matures early in life (which would favour a high value of r) but doesn't live quite as long as the North American grouper species (which would favour lower values of r). Assuming that these effects roughly cancel each other out, one would expect coral trout *a priori* to have a broadly similar value of r to the North American species.

1.8 Data used in the assessment

The following data are used in the remaining chapters of this report:

- Scientific, fishery-independent measures of fish abundance during and after replenishment closures of Boult Reef (1983–1986) and Bramble Reef (1992–1995) (see Table 1). These data are only discussed qualitatively in Chapter 2, and are not analysed further, although they justify the inclusion of the naivety parameter in the population dynamic model (see Chapter 6).
- Data on tropical cyclones from two separate databases: the Best Tracks database maintained by the Joint Typhoon Warning Center (JTWC), an agency of the United States Department of Defense; and the tropical cyclone database maintained by the Australian Bureau of Meteorology (BoM). The cyclone data of most interest were those relating to time, location, size and wind speed (see Chapter 3).

- Fishery-dependent commercial logbook data from the database maintained by Fisheries Queensland, 1988–2013 (see Chapter 4). Standardised catch rates, adjusted for naivety of the fish population, were used as relative measures of abundance to compare one year to another (not to provide an absolute number of fish per hectare).
- Fishery-dependent recreational catch size data estimated from diary surveys conducted by Fisheries Queensland in 1997, 1999, 2002, 2005 and 2011 (Higgs, 1999; 2001; Higgs et al., 2007; Taylor et al., 2012), and the National Recreational and Indigenous Fishing Survey carried out in 2000 (Henry and Lyle, 2003).

- Fishery-independent estimates of abundance of coral trout from underwater visual surveys funded by GBRMPA: a major survey of hundreds of reefs was undertaken from 1983 to 1986 (see Ayling and Ayling, 1986), and many smaller-scale surveys were carried out in later years. Surveys conducted by the divers Tony and Avril Ayling were used as absolute measurements of abundance of coral trout in numbers of fish per hectare (see Chapter 5).
- Fishery-independent age-frequencies and underwater visual survey abundances of coral trout obtained by the Effects of Line Fishing (ELF) Project, a major research project run by CRC Reef Research Centre and partly funded by the Fisheries Research and Development Corporation (see Mapstone et al., 2004b). The ELF Project ran from 1995 to 2005 and sampled 24 reefs in the GBR (four clusters each of six reefs) each year. It had special permission from GBRMPA to sample in green zones, which produced some very valuable data. In addition, some of the reefs had their zoning changed during the course of the project, in order to provide extra information on the effects of fishing.

- Fishery-independent age frequencies collected on reefs off Townsville by James Cook University in the early 1990s, provided by Prof. Garry Russ (described by Russ et al., 1995)
- Fishery-independent age frequencies collected by Fisheries Queensland 2006–2009 from four clusters of reefs in the GBR.
- Fishery-independent underwater visual survey data collected by the Australian Institute of Marine Science's Long Term Monitoring Program (AIMS LTMP), 1992–2011, used as relative measures of abundance, provided by Dr Hugh Sweatman.
- Fishery-independent underwater visual survey data collected by Fisheries Queensland, 1999–2002, used as relative measures of abundance.
- Fishery-independent underwater visual survey data on near-shore reefs collected by James Cook University, provided by Dr David Williamson: Keppel Islands region 2011, Palm Island and Whitsunday Islands regions 2009. These data provided information on the species splits between *Plectropomus maculatus* and *P. leopardus* on near-shore reefs.

- Mapping data of the GBR provided in electronic form by GBRMPA, providing detailed maps of every reef in the GBR. The major data used were the maps of "dry reef" (emergent reef that protrudes above the high-water mark) and "wet reef" (dry reef plus submerged reef down to a depth of roughly 15–20 m). The principal habitat of coral trout was taken to be the wet reef that was not in the dry-reef map. The submerged reef deeper than the mapped wet reef probably also hosted a large population of coral trout which could not be quantified. The mapping database also provided the Bioregions upon which the regional structure of the stock assessment is based.
- Zoning records of every reef in the GBR, sourced from historical maps published by GBRMPA (see zoning dates in Table 1); current zoning (since 2004) was provided in electronic form by GBRMPA.

2. Reef replenishment closures

2.1 Boult Reef

Boult Reef (official number 23-079) and North Reef (23-045A), located in the Capricorn–Bunker Region of the Great Barrier Reef, were closed for replenishment of fish stocks on 1 July 1983. North Reef reopened on 1 July 1986, and Boult Reef reopened on 1 December 1986. The reopening of Boult Reef was intensively studied, but North Reef was not.

The Boult Reef opening experiment, which lasted 14 days, is documented in detail by Beinssen (1989). For the purposes of this stock assessment, his most important finding was that the catch rate of coral trout fell dramatically once fishing commenced on a population that had not been fished for several years. This is illustrated in Figure 11 and explained in the following paragraphs.

A short-term tagging experiment, in which 375 fish were tagged and 93 were returned, estimated the population size of legal-sized (≥ 35 cm) coral trout on Boult Reef at 8613 fish, with a standard error of 873 fish. This population estimate was confirmed by underwater visual surveys conducted as part of the Boult Reef experiment. The habitat on the reef was estimated at 342 ha, producing a population density estimate of about 25 adult fish per hectare, which is what might be expected from underwater visual surveys conducted throughout the GBR (Ayling and Ayling, 1986), considering that the Capricorn–Bunker reefs are at the southern end of the habitat range of coral trout. About 25% of the population was caught in the 14 days of the experiment.

If catch rate were proportional to abundance, one would expect a plot of catch rate against the cumulative number of fish removed to show a slow, linear decrease (dotted line in Figure 11). The observations diverge greatly from this expectation. Beinssen explained the discrepancy by a "feeding phase" hypothesis whereby coral trout went on and off the bite, and only those that were in the feeding phase at the time of reopening would take bait. This hypothesis was disproven by subsequent experiments (GBRMPA unpublished note to report by Beinssen, 1989), and it was not published in the peer-reviewed literature.

A second explanation could be hyperdepletion, whereby fishers first deplete a few locations of high abundance and then proceed to fish locations of lower abundance (Hilborn and Walters, 1992, pp. 190–191). Then the catch rate can fall sharply when the total population is scarcely depleted at all.

The hyperdepletion explanation requires a change in fisher behaviour over time, a conscious move to places where abundance is lower. No literature or personal communications by fishers of which we are aware have mentioned the possibility of such moves within a single reef. Our understanding is that a reef has numerous "hangs" on which relatively high catch rates are achieved, and fishers fish the same set of hangs whatever the level of the overall catch rate. A hang on which a fisher has been fishing often still has plenty of coral trout visible through a view bucket, but when they no longer bite the fisher has to move on. It may take some time to precisely locate the next hang, but we know of no systematic expectation among fishers that it will have fewer fish than the previous one.

With respect to stock assessment, we note that hyperdepletion gives the same outcome and model parameterisation as the social learning hypothesis outlined below. Therefore it does not matter in the end which of the two hypotheses is true.

The third explanation of the results on Boult Reef is the presence of social learning in coral trout populations, and is the only explanation of which we are aware that fits the facts. The social learning hypothesis is that coral trout learn from each other by some means (e.g., direct observation of each other, or a chemosensory cue), and the fish that are caught highlight the danger to other fish who then modify their behaviour to avoid taking bait (see discussion of social learning in section 1.6).

Social learning can be modelled by making the catch rate proportional to the population density of a "shadow" population of fish that endures a fishing mortality rate several times higher than the actual fishing mortality rate. In the context of the short-term Boult Reef experiment, this social learning model states that the catch rate is proportional to the depleted population density raised to some power greater than 1. This model was fitted to Beinssen's data, which are listed in Table 3.

The social learning model fits the data in Figure 11 tolerably well, although the figure suggests that the catchability (ratio of catch rate to population density) of coral trout may level out at about one quarter of the level of a virgin population. The estimate of the social learning parameter is about 7.5, i.e., the catch rate fell as if the fishing intensity were about 7.5 times its actual value, which is a very large adjustment. More experimentation would be required to determine whether the catchability really levels out or continues to fall. The social learning model was fitted in the statistical software *R* (R Core Team, 2013), using the following code:

```
CumCatch = sqrt(cumsum(Catch) * (cumsum(Catch) - Catch))
PopSize = 8613
Surv = 1 - CumCatch / PopSize
lf = glm(Catch ~ log(Surv), family = quasipoisson(link = "log"),
 offset = log(Effort))
```

The social learning model is described in more detail in section 6.8 later in this report.

It is also notable from Beinssen (1989) that the coral trout on Boult Reef at reopening and for some years afterwards were substantially bigger than those on neighbouring reefs, so there is little doubt that this closure was respected by all, or very nearly all, of the fishers on the GBR. Enforcement of the closure would have been helped by the fact that the Capricorn–Bunker reefs were a popular tourist destination. Such effectiveness of closures in areas frequented by tourists has also been observed in recent years in the inshore Keppel Island reefs (David Williamson, James Cook University, personal communication).

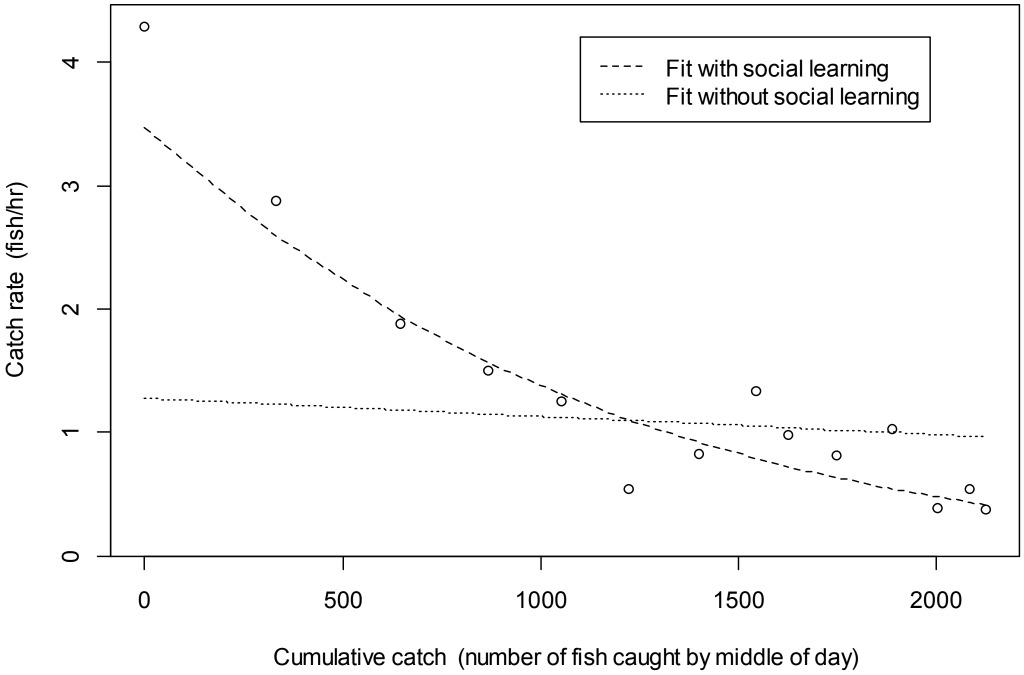


Figure 11: Catch rate data of Beinssen (1989), showing that catch rate falls at a much steeper rate than would be expected from the number of fish removed. A model that takes account of social learning fits much better than the one that does not.

Table 3: Catch and effort data from Beinssen (1989).

Day	Catch (number of fish)	Effort (hours)
1	207	48.25
2	322	111.75
3	255	136.00
4	179	119.50
5	185	148.25
6	154	286.25
7	201	242.50
8	85	63.50
9	78	79.50
10	162	198.25
11	123	120.00
12	102	267.00
13	60	111.00
14	23	62.00

2.2 Bramble Reef

Bramble Reef (18-029), fairly close to Townsville, was closed for replenishment on 1 January 1992, and reopened on 1 July 1995. Many underwater visual surveys, structured line-fishing surveys and fishing fleet surveys were conducted on Bramble Reef and neighbouring "control" reefs that were open to fishing, before and during the closure and after the reopening, making this probably the most intensively studied fishery management event in the history of the GBR. The story of the closure is recounted by Robertson et al. (1998). The opening of the reef was widely advertised, and the fishing intensity after opening was very high. Many commercial boats came from other parts of the GBR to fish the Bramble Reef reopening, and also increased the fishing pressure on other reefs in the area.

The Bramble Reef experiment was confounded by two factors that were not present in the Boult Reef closure:

- It coincided with recruitment to the fishery of an extremely strong year class of coral trout, which distorted the catch rates and suppressed the recruitment of subsequent year classes (Russ et al., 1995; 1996) (see discussion in section 1.5.2).
- There is strong evidence that Bramble Reef was not effectively closed to fishing during the closure period. The observed numbers of fishing vessels on Bramble Reef after it was closed were about the same before as before it was closed, and were higher than the numbers on five of the six control reefs (Ayling and Ayling, 1997, section 3.6).

Nevertheless, the intensity of fishing immediately after reopening was probably the highest ever applied to a reef on the GBR, so the experiment should still provide some valuable information.

Underwater visual surveys conducted in May 1995, shortly before the reopening, showed that the density of adult coral trout on Bramble Reef was about 2.2 times that on the control reefs (Ayling and Ayling, 1997). Previous surveys during the closure had found no significant difference in abundance between Bramble Reef and the control reefs.

Ayling and Ayling (1997) note that closer inspection of the data indicated that the higher density was due mainly to higher recruitment on Bramble Reef than on neighbouring reefs over the preceding few years. Given that (a) spawning stocks were not significantly different from the control reefs until the May 1995 survey, (b) Bramble reef was not truly closed to fishing, and (c) there was little or no evidence that spawning stocks on the control reefs were dangerously low, the higher recruitment on Bramble Reef was unlikely to have been caused by the closure to fishing. The higher abundance of coral trout on Bramble Reef was also associated with a higher abundance of forage (prey) fish, primarily Pomacentrids, which again would not be related to the closure.

The surveys found no appreciable difference in fish length between Bramble Reef and the control reefs during the closure, which supports the contention that the reef was not effectively closed to fishing.

Post-opening underwater visual surveys carried out in August 1995 showed that about 60% of the population of adult coral trout on Bramble Reef had been removed by fishing in the eight weeks since the reef opened to fishing on 1 July 1995 (Ayling and Ayling, 1997), which reduced the population density on Bramble Reef to that on the control reefs. The fishing effort at Bramble Reef on the day of opening was very high: 64 recreational fishers and between 26 and 40 commercial fishers from 14 primary vessels (Mapstone et al., 1996). Mapstone et al. also comment that the level of fishing on Bramble and neighbouring reefs was negligible between late July and September 1995, due to persistent strong winds. Therefore the depletion must have taken place in just a few weeks in July 1995.

Habitat calculations combined with underwater visual surveys estimated the pre-opening population of adult fish on Bramble Reef as 20,000, 12,000 of which were then caught within a few weeks of opening (Ayling and Ayling, 1997). Ayling and Ayling comment that such levels of fishing are abnormally high, produced by advertising and the level of interest in the opening, and are not experienced anywhere on the GBR under normal conditions.

Mapstone et al. (1996) state that catch rates from research fishing were about twice as high on Bramble Reef as on the control reefs immediately prior to opening, but fell to about the same rate as the control reefs within two weeks.

Data from voluntary logbooks maintained by the commercial fleet as part of the experiment showed a fall in the catch rate after a few days (Mapstone et al., 1996; Davies and Mapstone, 2012), but not greatly in excess of what would be expected from the population depletion. A suitable value for the social learning parameter (see description in section 2.1 above) was about 1.5 (data from Davies and Mapstone, 2012), much less than the value of 7.5 estimated for the Boult Reef experiment.

The low indicated value of the social learning parameter supports the hypothesis that catchability levels out as fishing continues. Social learning had only a small effect on Bramble reef because Bramble reef was already being fished at a moderate level when the experiment began.

Social learning can therefore be viewed as a “naivety” effect, whereby a fish population that has not been fished for a long time is naïve and its members are easily caught. After about 25% of the population has been removed, the naivety disappears and the catchability settles down at about one-quarter of its initial value. This gives rise to a relationship between catch rate and population depletion such as that plotted in Figure 12.

Another interesting finding by Davies and Mapstone (2012) was a daily effect on catchability, whereby the catch rates were consistently higher in the morning than the evening on heavily fished reefs. They recovered overnight to be higher in the morning than they were the previous evening.

It appears that there may be different time scales for retention of knowledge by coral trout. Sometimes fish forget overnight, but other times they retain knowledge for weeks or months. This would be in accord with previous findings about learning, whereby fish retain knowledge only for as long as they think it will be beneficial (Brown et al., 2011b, section 4.6).

The density on both Bramble Reef and the control reefs continued to fall over the following year, as indicated by further underwater visual surveys conducted in May 1996 (Ayling and Ayling, 1998). This was probably because the strong year class (see above) became less abundant as it advanced in age, and the cohorts that replaced it were much weaker.

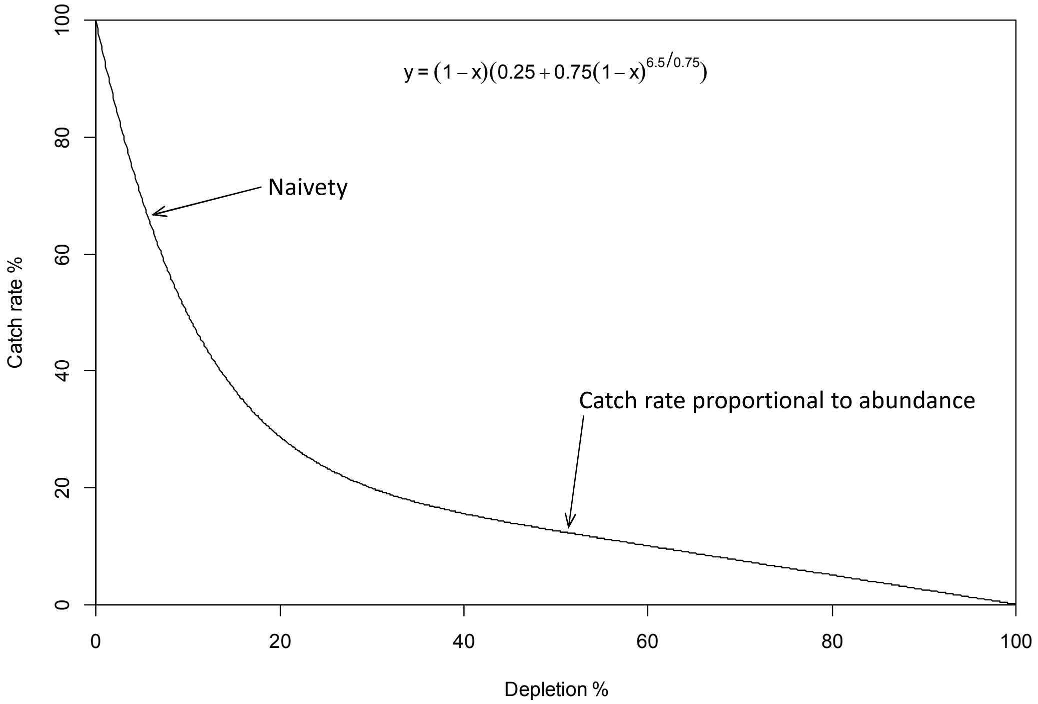


Figure 12: Idealised plot of catch rate against depletion ratio for coral trout, combining the results of the Boult Reef and Bramble Reef replenishment closures.

2.3 Information transferred into the stock assessment

The Boult Reef and Bramble Reef replenishment experiments demonstrated the need to model social learning (or, equivalently, hyperdepletion) in the stock assessment, and furnished the following knowledge to the stock assessment of coral trout:

- Catch rates fall much faster than the population is depleted from the unfished level.
- After the population size has been depleted by about 25%, the catchability (ratio of catch rate to population size) seems to stay constant at about 25% of the unfished level.
- Social learning is the most credible explanation of the fall in catch rates, but hyperdepletion would have the same effect and model formulation.
- The stock assessment model can attempt to estimate a social learning parameter from population size estimates and commercial catch rates (see section 6.8 later in this report).
- If the social learning parameter cannot be estimated sensibly within the model, the relationship shown in Figure 12 can be used instead. The social learning parameter can be fixed at 7.5 if the instantaneous fishing mortality rate F is less than about $0.6\,\mathrm{yr}^{-1}$ (the level that depletes the population by about 25% by midyear), while the catchability can be fixed at about 25% of the unfished level if $F > 0.6\,\mathrm{yr}^{-1}$. This formulation assumes a knowledge retention time of about six months for coral trout. No data are available on the actual retention time.

3. Tropical cyclones

3.1 Background

Tropical cyclones are known to have a big effect on the fishery on the Great Barrier Reef. Catch rates of coral trout fall after a cyclone (Tobin et al., 2010), whereas catch rates of red throat emperor (*Lethrinus miniatus*) increase (Leigh et al., 2006). The effects are much longer-lasting than the oceanographic disturbances caused by tropical cyclones. Disturbances such as sea-surface temperature (SST) last a few weeks at most, but the effects on catch rates may last for years.

The following causal mechanisms have been suggested for the long-lasting changes in catch rates:

- For coral trout, physical damage to coral from a tropical cyclone takes away hiding places for forage fish, making it easier for predators such as coral trout to catch them. With increased availability of prey, coral trout become well-fed and less inclined to take dead bait until either the coral grows back or they deplete the supply of prey (suggested by commercial fishers, personal communications).
- For red throat emperor, the ocean mixing associated with a tropical cyclone brings up younger red throat emperor from deeper water into shallower water where the fishery operates. Once recruited to shallower waters, they stay there for the rest of the lives. Cyclones thereby cause premature recruitment of red throat emperor to the fishery (suggested by Leigh et al., 2006).

These mechanisms have not been verified by scientific studies, and are offered only as *possible* explanations for the observed effects on catch rates and, in the case of red throat emperor, age frequency distributions which showed an influx of young fish to the fishery after Cyclone Justin.

We note that the presence of predators has been shown to have a big influence on how far forage fish venture from the coral structures that provide them with cover (Madin et al., 2010; 2011). Cover is evidently important to the survival of forage fish, which provides support to the coral trout hypothesis above.

The hypothesis that coral trout may be better able to feed after a tropical cyclone raises the possibility that cyclones may indirectly cause a depletion of forage-fish stocks a few years later, which could adversely affect the population of coral trout, making them hungry and more willing to take bait. This may result in quite complex cycles of many years' duration in coral trout catch rates, which would have little relationship to coral trout abundance.

It is apparent from observations of cyclones and catch rates that the rare large cyclones that remain close to the Queensland east coast for some days have a much bigger effect than those that come in from the ocean, strike land immediately, and dissipate soon afterwards. Commercial fishers stated that Cyclone Justin in 1997, which stayed off the coast for two weeks, had by far the biggest effect on the fishery of any cyclone since the introduction of logbooks in 1988, until Cyclone Hamish struck in 2009. Justin was found to have such a big effect on catch rates of red throat emperor that it required special modelling as a one-off event (Leigh et al., 2006). Justin was extremely large in size but relatively weak in strength. Hamish was a very powerful cyclone that moved parallel to the coast and had a huge effect on the coral trout fishery (Tobin et al., 2010).

The tracks of Cyclones Justin and Hamish are shown in Figure 13, along with that of Cyclone Yasi in 2011 which followed the typical pattern of coming from the east, striking land and quickly dissipating. Yasi was, however, one of the largest and most powerful cyclones in Queensland's recorded history, and caused severe damage to reefs between Cairns and Townsville (Ayling and Ayling, 2011; GBRMPA, 2011). A satellite image of Yasi is shown in Figure 14.

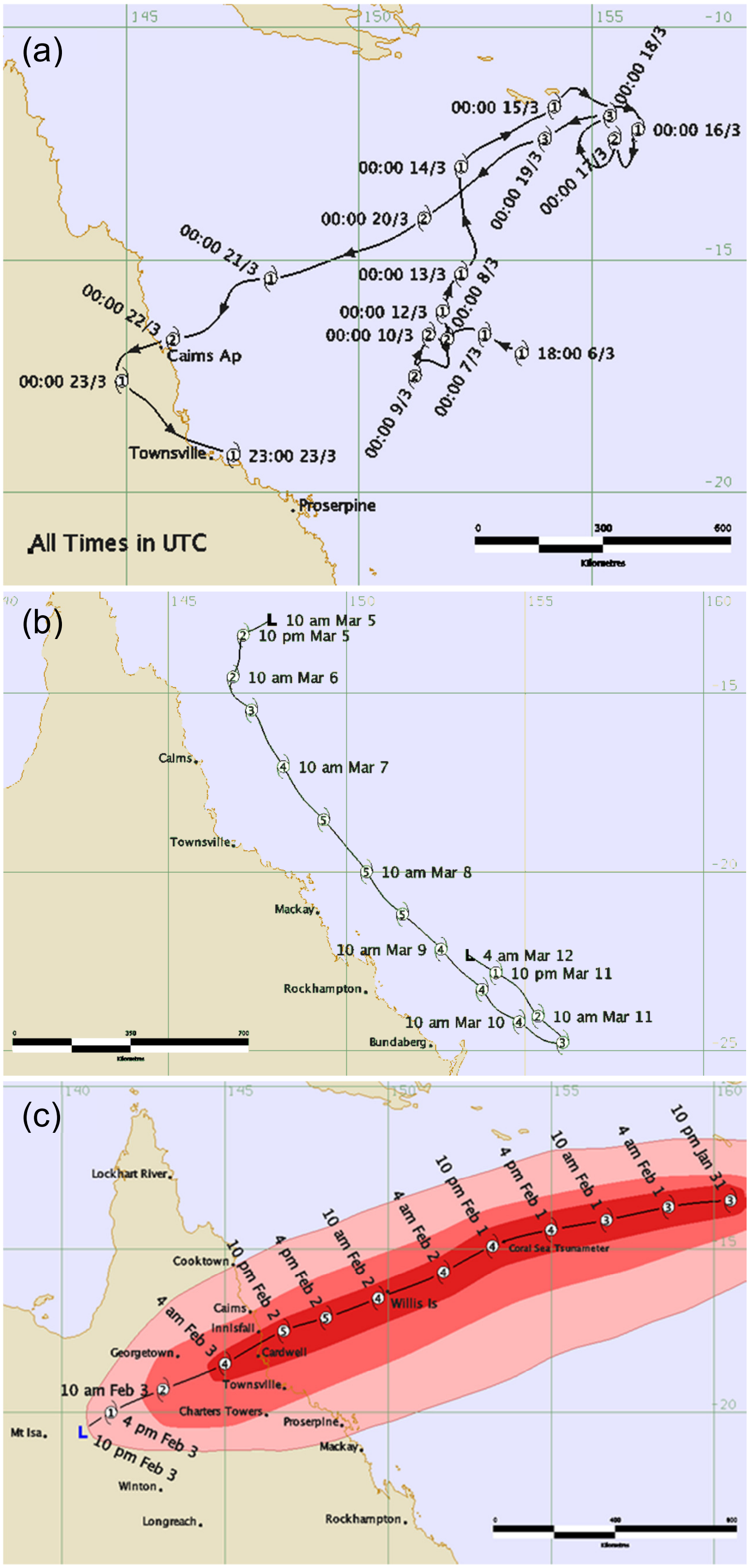


Figure 13: Tracks of three major tropical cyclones: (a) Justin, March 1997; (b) Hamish, March 2009; (c) Yasi, February 2011. The numbers shown in circles are the Australian cyclone categories, ranging from 1 (weakest) to 5 (strongest). Source: Australian Bureau of Meteorology, www.bom.gov.au, Creative Commons by Attribution licence.

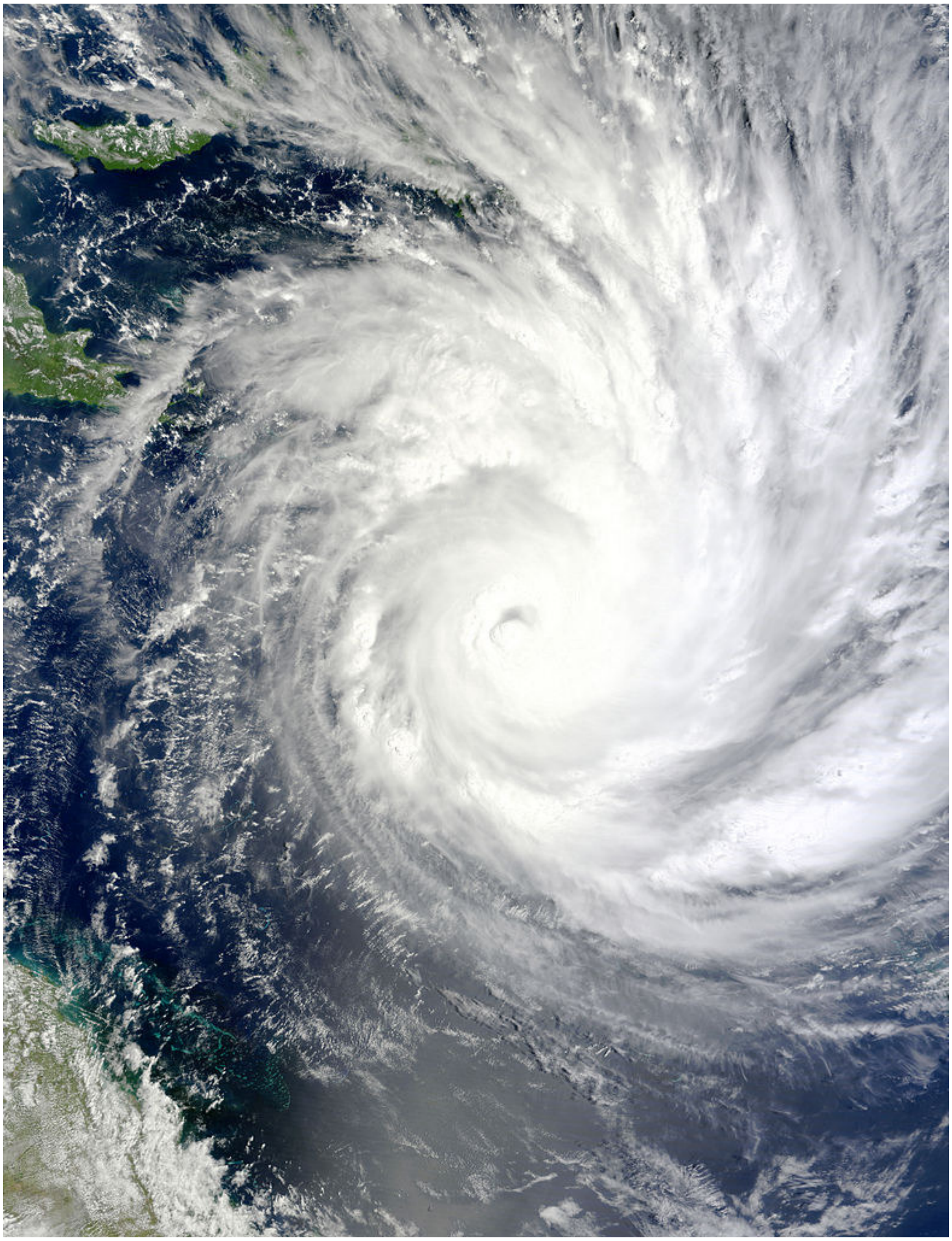


Figure 14: Satellite image of Severe Tropical Cyclone Yasi, with the Queensland coast and southern GBR on the bottom left and Papua New Guinea at the top left. Source: NASA image by Jeff Schmaltz, MODIS Rapid Response Team at NASA GSFC, http://rapidfire.sci.gsfc.nasa.gov, licence similar to Creative Commons by Attribution.

This chapter presents an analysis of wind energy data on tropical cyclones, for the purpose of improving the standardisation of commercial catch rates as an index of abundance. The underlying assumption is that the fall in coral trout catch rates following a cyclone generally does not correspond to a fall in abundance of coral trout; this has been confirmed by underwater visual surveys (Tobin et al., 2010).

Wind energy was chosen because it was available for most cyclones since 1988 (when Queensland commercial logbooks were introduced), had a direct effect on ocean mixing and habitat disturbance, and could be integrated over time to capture the known large effect of long-lived cyclones such as Justin and Hamish, compared to the many short-lived cyclones that strike the Great Barrier Reef. Other measures such as sea surface temperature may not capture this effect. More sophisticated inputs than raw wind energy are being pursued in another research project funded by the Australian Government's Fisheries Research and Development Corporation and the Great Barrier Reef Marine Park Authority (FRDC Project No. 2013/020).

3.2 Data sources

We use tropical cyclone records only from 1988 onwards. We note that climate-change scientists have used records beginning in the 1950s to look for changes in the frequency of cyclones, but the utility of that practice has been questioned due to improvements in cyclone observation and recording over that time (see Sriver and Huber, 2007). Modelling of cyclones has improved dramatically over the years, and we regarded records prior to 1988 as not useful for estimating wind speeds at fishing locations. We also observed a marked increase in the quality of records over the period from 1988 to 2003.

Modelling of tropical cyclones currently requires skilled human judgement, and in general is still not amenable to automated computer modelling. The major databases of cyclones are still assembled from individual expert judgements of particular cyclones (information kindly provided by Jeff Kepert of the Australian Bureau of Meteorology, personal communication).

Two major databases of tropical cyclones were available: the Best Tracks database maintained by the Joint Typhoon Warning Center (JTWC, an agency of the United States Department of Defense) and the Tropical Cyclone Database maintained by the Australian Bureau of Meteorology (BoM). The two databases often differ markedly prior to 2003, but largely agree from that year onwards. We merged the two databases to form a combined database.

A notable difference between the two databases is that the JTWC database uses one-minute average wind speeds, whereas the BoM database uses ten-minute average wind speeds. Hence JTWC maximum wind speeds tend to be higher (JTWC, 2012; BoM, 2014). A rough conversion factor is 0.871 (BoM, 2014), which we used (in the denominator) to convert the BoM wind speeds to an equivalent JTWC wind speed when the wind speed was not available in the JTWC database.

Our judgement was that the JTWC database relied on long-established methods and was less adventurous than the BoM one, and we have generally preferred the JTWC fields over the BoM ones in our analysis. The JTWC database, however, was often out of date by up to two years, whereas the BoM database was impressively kept right up to date. BoM records were the only ones available for recent cyclones.

An example of the differences between the two databases is that Cyclones Steve (2000), Tessi (2000), Vaughan (2000) and Grace (2004) were given much higher maximum wind speeds (over hurricane force) in the BoM database (42, 39, 47 and $25.7\,\mathrm{ms}^{-1}$) than in the JTWC database (33, 26, 26 and $18\,\mathrm{ms}^{-1}$), whereas they should have been lower due to the ten-minute versus one-minute averaging. The BoM database does not contain any associated radii of hurricane force or (in the case of Grace) storm force winds, which it should do if the maximum wind speeds are correct. The central atmospheric pressures in the BoM database, although only a rough guide, support lower wind speeds as in the JTWC database. BoM itself states that Steve, Tessi and Vaughan never reached "severe" status (Category 3 or above, defined as hurricane force winds, $33\,\mathrm{ms}^{-1}$) (BoM, 2000a; b; c).

A notable instance of a JTWC data value not being credible is the value of 879 hPa as the minimum central atmospheric pressure reached by Cyclone Monica in 2006 while it was in the Gulf of Carpentaria, which if true would be one of the lowest values ever found for a tropical cyclone anywhere in the world. The BoM database gives the value as 916 hPa (although updated methodology used by BoM more recently would give 905 hPa). This error was regarded as an isolated instance, arising from applying standard methodology to an unusual cyclone. The maximum wind speed for Monica is consistent between the two databases (80 ms^{-1} one-minute average in JTWC, 69 ms^{-1} ten-minute average in BoM).

The combined database generally had records of a cyclone's location, central atmospheric pressure, maximum wind speed and radii of gale force (17 ms^{-1}), storm force (25 ms^{-1}) and hurricane force (33 ms^{-1}) winds every six hours. In later years the records of wind speed radii were available by quadrant (northeast, southeast, southwest and northwest).

3.3 Analysis

3.3.1 Wind speed as a function of distance from centre

Analysis of the wind speed data was difficult and often counter-intuitive. As a cyclone intensified, it generally became larger in radius, and the maximum wind speed increased, but the radius of maximum wind speed, on the other hand, decreased.

On the same theme of decreasing radius, it is recognised that "midget" cyclones are much more destructive than larger cyclones with the same central pressure, although the affected area is of course much smaller (Callaghan, 2014). Notable midget cyclones to hit the GBR are Ada which hit the Whitsunday Islands in 1970, and Larry which hit the town of Innisfail in 2006. Also Cyclone Tracy which devastated the city of Darwin in the Northern Territory in 1974 was a midget.

There appeared to be a qualitative difference between cyclones that reached hurricane force wind speed during their lifetimes and those that did not. To attain hurricane force, a cyclone apparently had to follow a certain pattern, whereas cyclones that did not attain hurricane force during their lifetimes showed much more diversity.

The databases also contained many inconsistencies that had to be resolved manually. For example, sometimes maximum wind speed was above gale force, storm force or hurricane force but the corresponding radii of winds of that speed were not provided, implying that that force was not reached. Also sometimes the cyclone radii changed greatly from one six-hour measurement to the next, which was not realistic.

The following decisions were made during the analysis:

- Wind speeds below gale force were ignored, because they could arise from meteorological events that were not classified as tropical cyclones.
- Wind speed was set equal to the maximum wind speed where the distance from the centre was between the eye radius and the radius of maximum wind speed.
- Wind speed inside the eye was assumed to be below gale force, and was ignored.
- When eye radius was not available, it was set according to the formula (*R* code)

  ```
  Eye = pmax(EyeMin, pmin(EyeMax, EyeRat * MRD))
  ```

  where `EyeMin` (minimum eye radius) was set equal to 4.63 km, `EyeMax` (maximum eye radius) was set to 37.04 km, and `EyeRat` (average ratio of eye radius to radius of maximum wind speed) was set to 0.5087015 (estimated from a regression using data for which eye radius was available); `MRD` denoted the radius of maximum wind speed.
- When `MRD` was not available it was estimated from the maximum wind speed `Vmax` which was always available. The estimation was by a linear regression with a change of slope at hurricane force:

  ```
  MRDPred = 39.9116075 - 0.1578765 * Vmax - 2.0853134 * VmaxSubH
  ```

  where

  ```
  VmaxSubH = (Vmax <= Speed_HF) * (Vmax - Speed_HF)
  ```

and `Speed_HF` was equal to hurricane force speed, 33 ms^{-1}. Wind speeds were measured in metres per second, and distances in kilometres.

- When radius of gale force wind was not available it was estimated from a regression that included a linear term in `Vmax`, linear and quadratic terms in `VmaxAll`, and a linear term in `MRD` when `MRD` was available; different regressions were used according to whether `MRD` was available. The radius of gale force wind was also constrained to be greater than or equal to either the actual value of `MRD` or the predicted value of `MRD` from above. The variable `VmaxAll` was equal to the maximum value of `Vmax` over the lifetime of the cyclone. Coefficients for the case where MRD was available were

  ```
  (Intercept)  3.362759036
  Vmax         0.016743324
  VmaxAll      0.034354816
  VmaxAllSq   -0.000284172
  MRD          0.004275399
  ```

  and for the case where `MRD` was not available they were

  ```
  (Intercept)  3.851422003
  Vmax         0.011848155
  VmaxAll      0.036463726
  VmaxAllSq   -0.000335066.
  ```

The most important part of the analysis was the interpolation of wind speed as a function of distance from the centre of a cyclone, at distances beyond the radius of maximum wind speed. This was fitted by the following relationship:

`(v - v0) / (vm - v0) = ((d0 - d) / (d0 - dm))^alpha` (3.1)

where `v` denotes wind speed, `d` is the square root of distance, `v0` and `d0` are the values of `v` and `d` at gale force, and `vm` and `dm` are the values at the cyclone's maximum wind speed. The parameter `alpha` was estimated at `2.112458`, fitted by nonlinear least squares with `d` as a function of `v`. When radii of storm force and hurricane force winds were available, they were used in the interpolation when appropriate by altering the values of `v0` and `vm` to exactly match these additional data points. For example, if both storm force and hurricane force radii were available, wind speeds at distances in between them were estimated from (3.1) by first altering `v0` and `vm` so that (3.1) produced a value `v` of storm force wind speed (25 ms^{-1}) at the storm force distance, and hurricane force wind speed (33 ms^{-1}) at the hurricane force distance.

3.3.2 Wind energy by fishery grid square

Wind energy transfer into the sea per unit of time was assumed to be proportional to the cube of wind speed (Bister and Emanuel, 1998; Emanuel, 1999), on the basis that the kinetic energy of the air is proportional to the square of wind speed and then the rate of air passing a given point on the water is proportional to the wind speed: multiplying these two powers provides the cubic relationship.

Conversion to standard units of energy (petajoules) was not necessary for subsequent analysis of fishery catch rates, because only relative cyclone strengths were required, but we did it out of interest, to allow comparison to the energy used in everyday processes. The parameter values used were 0.002 for the surface drag coefficient (dimensionless), and 1 kg m^{-3} for the surface density of air (Sriver et al., 2008).

The above assumptions allowed the total wind energy transferred into each six-nautical-mile fishery grid square by each cyclone to be calculated. The monthly total energy input to each grid square was a measure of fishery disturbance, for later use in catch rate standardisation.

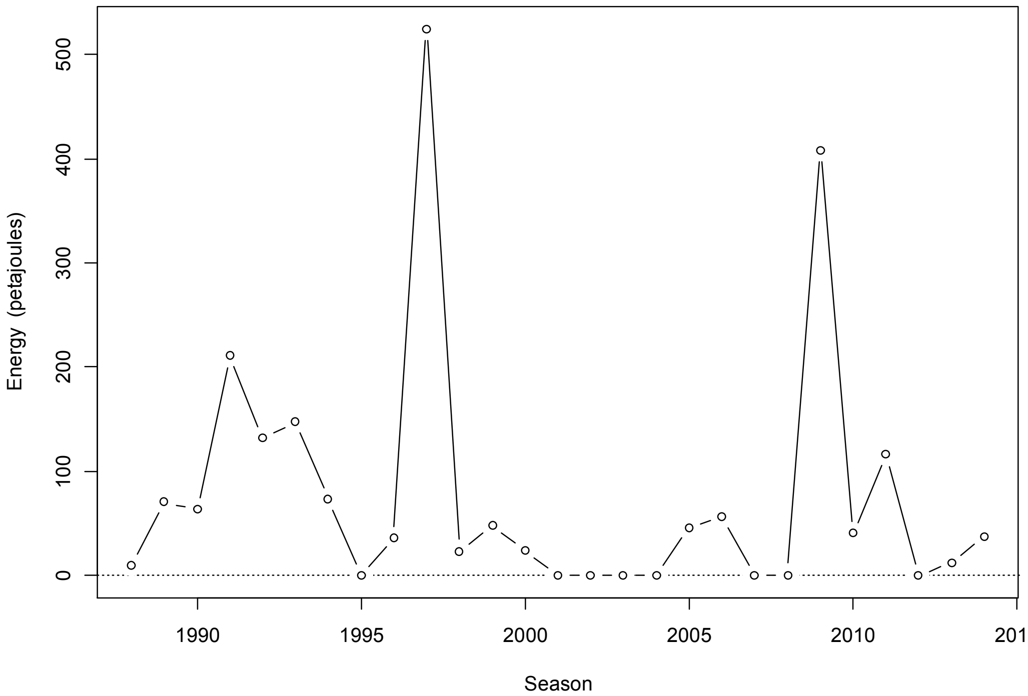


Figure 15: Tropical cyclone energy inputs to the coral trout fishery since logbooks began, by cyclone season (e.g., 2011 denotes the 2010–11 season). The two peaks caused by Cyclones Justin (1997) and Hamish (2009) are notable, as are the substantial amount of cyclone activity up until 1994 and the low level of cyclone activity from 1998 to 2008.

3.4 Results and discussion

Cyclone energy inputs by cyclone season are plotted in Figure 15. The two cyclones with the greatest energy inputs were Justin in 1997 and Hamish in 2009, which produce the twin peaks in Figure 15. It is notable that no cyclones with energy inputs greater than 50 PJ struck the GBR between these two cyclones, although there were many up until 1994.

Energy inputs from individual tropical cyclones to hit the Great Barrier Reef are listed in Table 4. Energy inputs specific to fishery grid squares were also calculated and stored for use in the catch rate standardisation in Chapter 4.

Notable tropical cyclones to affect the GBR between 1962 and 1987 are listed in Table 5. We have not attempted to calculate energy inputs for these cyclones, due to inferior cyclone modelling methodology in use at that time. The table does not list minor cyclones.

The fishery for live coral trout may have developed over an exceptionally favourable period of low tropical cyclone activity. It may not be realistic to base business plans on a return to the conditions of the early 2000s.

The two recent extremely destructive cyclones, Hamish and Yasi, appear to be exceptions even by the standards of the early 1990s. The report by GBRMPA (2011) on damage caused by Yasi notes (p. 13), “A major difference between the impacts of TC Yasi and most other cyclones is the sheer scale of damage. Most other studies have reported damage spanning less than 200 km of reef; damage from TC Yasi stretched over 400 km. The only other cyclone known to have caused damage on a similar spatial scale is TC Hamish, which travelled along the southern Great Barrier Reef in 2009.” Yasi was extremely large and powerful, although it spent only a short time over the GBR, which is why its energy input is much less than that of Hamish.

Table 4: Cyclones with winds of gale force (17 ms^{-1}) or above to affect the Great Barrier Reef since 1988. Location is the part of the GBR most affected, roughly classified as North (Cape York to Cairns Subregions), Central (Townsville Subregion and Whitsunday Islands) or South (Mackay Region south of the Whitsundays, to Swains or Capricorn–Bunker) Pmin = minimum central pressure (hPa); Vmax = maximum wind speed (ms^{-1}); Energy = estimated energy (PJ) input to ocean mixing by winds of gale force or above in locations in the Reef Line Fishery. Cyclones with mixing energy of 50 PJ or more are listed in bold, while those less than 5 PJ are in grey. Note that Pmin and Vmax are taken over the life of a cyclone, including times when it is not impacting the Great Barrier Reef.

Name	Dates	Location	Pmin	Vmax	Energy
Charlie	21–29 Feb 1988	Central	972	21	10.0
Aivu	**01–04 Apr 1989**	**Central**	**935**	**62**	**70.4**
Felicity	15–18 Dec 1989	North	975	31	4.2
Hilda	04–08 Mar 1990	South	970	31	1.0
Ivor	**16–22 Mar 1990**	**North**	**965**	**39**	**58.0**
Joy	**19–26 Dec 1990**	**Central**	**940**	**46**	**202.8**
Kelvin	25 Feb – 5 Mar 1991	North	980	28	8.2
#18	20 Feb 1992	South	NA	18	2.6
Betsy	06–15 Jan 1992	South	949	49	0.1
Mark	08–10 Jan 1992	North	980	28	1.2
Fran	**06–17 Mar 1992**	**South**	**898**	**72**	**127.8**
Nina	23 Dec 1992 – 04 Jan 1993	North	960	39	8.5
Oliver	**04–12 Feb 1993**	**South**	**950**	**59**	**53.0**
Roger	**12–22 Mar 1993**	**South**	**980**	**28**	**85.8**
Rewa	**28 Dec 1993 – 21 Jan 1994**	**South**	**920**	**64**	**73.6**
Celeste	26–29 Jan 1996	Central	965	33	25.5
Dennis	13–17 Feb 1996	North	990	23	2.0
Ethel	08–13 Mar 1996	North	980	23	8.3
Gillian	10–12 Feb 1997	Central	995	23	1.9
Ita	24 Feb 1997	Central	994	18	3.6
Justin	**06–25 Mar 1997**	**Central**	**974**	**55**	**519.2**
Katrina	02–24 Jan 1998	North	940	46	0.2
Nathan	21–30 Mar 1998	North	990	33	22.7
Rona	10–12 Feb 1999	North	970	28	47.7
Steve	26 Feb – 01 Mar 2000	North	975	23	3.7
Tessi	01–02 Apr 2000	Central	980	26	18.4
Vaughan	03–06 Apr 2000	North	975	26	1.6
Grace	21–22 Mar 2004	South	997	18	0.2
Ingrid	06–12 Mar 2005	North	910	67	45.6
Larry	18–20 Mar 2006	North	937	59	16.9
Monica	17–24 Apr 2006	North	916	80	39.4
Guba	13–19 Nov 2007	North	967	39	0.3
Ellie	31 Jan – 01 Feb 2009	North	996	18	6.5
Hamish	**05–11 Mar 2009**	**South**	**922**	**69**	**401.8**
Olga	22–30 Jan 2010	North	989	23	3.5
Ului	11–21 Mar 2010	Central	918	72	36.9
Tasha	24–25 Dec 2010	North	993	21	1.7
Anthony	23–30 Jan 2011	Central	989	28	5.3
Yasi	**30 Jan – 03 Feb 2011**	**Central**	**922**	**69**	**109.0**
Tim	13–19 Mar 2013	South	984	26	8.6
Zane	29 Apr – 01 May 2013	North	983	31	3.5
Dylan	30 Jan 2014	Central	975	28	35.6
Edna	31 Jan – 05 Feb 2014	South	992	21	1.6

Table 5: Substantial tropical cyclones to affect the Great Barrier Reef, 1962–1987. Cyclones for which large swells were reported close to the GBR are listed in bold. Pmin = minimum central atmospheric pressure (hPa); Vmax = maximum wind speed (ms^{-1}). Sources: Australian Bureau of Meteorology (BoM) tropical cyclone database (for Pmin and comments), and Joint Typhoon Warning Centre Best Tracks database (for Vmax).

Name	Dates	Pmin	Vmax	Comments
Dinah	**28–30 Jan 1967**	**945**	**46**	Large NE swells; severe damage Rockhampton to Grafton (NSW)
Ada	03–18 Jan 1970	962	36	Very small but direct hit on Whitsunday Islands; 13 deaths
Althea	19–29 Dec 1971	952	33	Direct hit on Townsville
Emily	**27 Mar – 02 Apr 1972**	**942**	**33**	Huge seas; weakened before striking coast SE of Gladstone; 8 deaths at sea
David	**13–19 Jan 1976**	**961**	**NA**	Huge size; huge swells added to large tides; gales from Papua New Guinea to Lord Howe Island; struck coast S of Mackay
Colin	25 Feb – 04 Mar 1976	954	NA	Large waves in S Queensland; well offshore passing GBR
Simon	**22–22 Feb 1980**	**950**	**51**	Passed over Hard Line & Swains and struck coast, followed it SE and intensified; huge swells in Capricorn–Bunker Region
Freda	**24 Feb – 05 Mar 1981**	**962** (972)	**39** (31)	Very rough seas; moved parallel to coast, about 380 km offshore; parentheses are for time near Qld
Elinor	11 Feb – 03 Mar 1983	935	51	Weakened before approaching coast; max intensity 600 km offshore; long-lived in Coral Sea
Pierre	18–24 Feb 1985	986	23	Weak but moved parallel to and close to the coast for 3 days
Winifred	28 Jan – 02 Feb 1986	957	46	Intensified only just before striking coast close to Innisfail; serious damage on land

Hamish, although not exceptionally large in diameter (see below), was extremely powerful with a maximum wind speed of 69 ms^{-1} and a minimum central pressure of 922 hPa (see Table 4). Importantly and unusually, it moved almost exactly parallel to the Queensland coast for some days and affected much of the length of the GBR (see Figure 13(b)).

It may well be the case that the size of the swell generated by a tropical cyclone is more important than the wind energy input at a particular location (suggested by climate researcher Marji Puotinen, personal communication). The swell can be generated by winds hundreds of kilometres away, which would raise the importance of cyclones of large diameter. A notable example is Cyclone Roger which did not contribute a great deal of direct wind energy in Table 4 but may still have had a big impact (see Figure 9). It had a maximum radius of 650 km in the BoM database, which is comparable to that of Justin (740 km) and larger than Hamish (269 km) and Yasi (500 km). Further research is in progress to help resolve this matter (see section 3.1 above).

3.5 Information transferred into the stock assessment

The tropical cyclone analysis provided cyclone energy inputs to ocean mixing, in the form of cumulative energy inputs for the previous 12, 24, 36 and 48 months over each year-month combination and six-nautical-mile fishery grid square. These energy inputs were used in the catch rate standardisation in Chapter 4, under the assumption that cyclones made coral trout more difficult to catch for some time but did not affect their abundance.

4. Harvest sizes and abundance indices

4.1 Species split on inner-shelf islands

Inner-continental-shelf regions are often claimed to be the domain of bar-cheek coral trout (*Plectropomus maculatus*) rather than common coral trout (*Plectropomus leopardus*). Some unpublished data to check this were kindly provided by Dr David Williamson of James Cook University. Densities of *Plectropomus* species were measured by underwater visual surveys on three groups of inner-shelf islands: Palm Island group between Cairns and Townsville, Whitsunday group between Townsville and Mackay, and Keppel group near Rockhampton.

The data are summarised in Table 6. It is evident that *P. maculatus* predominates in the Keppel group, but the species are split roughly 50-50 in the Palm group, and *P. leopardus* is actually the more common species in the Whitsunday group. As expected, there are few individuals of *P. laevis*, as this species prefers offshore reefs.

Despite the fairly high abundance of *P. leopardus* on these inner-shelf reefs, commercial fishers rarely fish these locations, probably because much of their time would go into catching *P. maculatus* which is less profitable for them and which they would probably not be able to distinguish prior to raising the fish out of the water.

It is likely, however, that commercial fishers frequently fished inner-shelf reefs before the development of the live-export fishery which concentrates on *P. leopardus*. This is especially likely up until 1976 when *Plectropomus* species had no minimum legal size and the commercial fishery stabilised at a low level before expanding rapidly from 1977 to 1991 after the introduction of a minimum legal size (see section 4.2 below).

Recreational fishers are generally happy to catch either *P. leopardus* or *P. maculatus*, and would catch both species at many inner-shelf locations.

For stock assessment, the direct use of these data is simply to justify the decision to exclude inner-shelf Bioregions from the stock assessment, and include only the offshore Bioregions. The stock assessment concentrates on the commercial fishery and on *P. leopardus*. Results in the remainder of this chapter are for all coral trout species combined.

Table 6: Underwater visual survey densities (fish per hectare) of different Plectropomus species observed on inner-shelf islands opposite the GBR. Source: Unpublished data from surveys carried out by James Cook University, project led by Dr David Williamson and Dr Garry Russ.

Group	Year	Island	P. leopardus	P. maculatus	P. laevis
Palm	2009	Curacoa	25.6	31.1	5.6
Palm	2009	Orpheus	42.2	43.3	1.1
Palm	2009	Pelorus	26.7	9.4	2.8
Whitsunday	2009	Black	36.7	23.3	0.0
Whitsunday	2009	Border	64.4	17.8	0.0
Whitsunday	2009	Dumbell	110.0	50.0	0.0
Whitsunday	2009	Hayman	51.1	11.1	2.2
Whitsunday	2009	Hayman—East	37.8	2.2	0.0
Whitsunday	2009	Hook	55.8	7.9	0.6
Whitsunday	2009	Hook Blue	20.0	10.7	0.0
Whitsunday	2009	Langford	6.7	33.3	0.0
Whitsunday	2009	Whitsunday	10.0	24.2	0.8
Keppel	2011	Great Keppel	0.0	118.5	0.0
Keppel	2011	Halfway	4.4	126.7	0.0
Keppel	2011	Miall	0.0	46.7	0.0
Keppel	2011	Middle	5.0	60.0	0.0
Keppel	2011	North Keppel	1.7	76.7	0.0

4.2 Commercial harvest size

4.2.1 Queensland Fish Board Data

Commercially caught fish were by law marketed through the Queensland Fish Board until 1981, and Fish Board annual records compiled by Halliday and Robins (2007) provide information about the harvest size up until then. These data are reproduced in Table 7.

Some illegal marketing of commercially-caught fish outside the Board was known to have taken place, but the recorded catch sizes by the Board are consistent with the survey undertaken independently by Hundloe (1985) (see Table 8). The average price per kilogram in Table 8 is reasonable and indicates that the Board data are not grossly underreported. The price increases with latitude; it is unknown whether this is a real effect of supply and demand or there may have been alternative, illegal marketing channels and consequent underreporting in the southern regions.

For input to the stock assessment population model, each reporting station had to be assigned to one of the Subregions shown in Figure 7, and the harvest had to be converted from financial years to calendar years. Reporting stations were assigned as follows:

- Cairns was assigned 75% to the Cooktown Region and 25% to the Cairns Subregion, informed by the early years of commercial logbook data which began in 1988.
- Innisfail was assigned 75% to the Cairns Subregion and 25% to the Townsville Subregion, again informed by early years of logbook data.
- Townsville and Home Hill were assigned to the Townsville Subregion.
- Bowen was assigned 50% to the Townsville Subregion and 50% to the Mackay Region, as Bowen was roughly on the Region boundary.
- Mackay was assigned to the Mackay Region.
- Rockhampton and Gladstone were assigned 80% to the Swains and 20% to the Capricorn–Bunker Region, informed by early years of logbook data.
- Catches marketed from Bundaberg south were assumed to have been taken outside the Great Barrier Reef Marine Park, and were excluded.

The conversion to calendar years was done on a simple 50-50 split, as the fishery was not strongly seasonal.

There was also the matter of species split. Commercial fishers currently target *Plectropomus leopardus* for the live fish trade, which developed only after the period covered by the Fish Board data. Therefore there may not have been a preference for *P. leopardus* during the Fish Board period, and a sizable amount of the harvest may have been *P. maculatus*. This effect was impossible to quantify. We assumed that all catches were *P. leopardus*, but, as will be described later, the stock assessment concentrates on locations that are currently frequented by commercial fishers.

The final assignment to Subregions is plotted in Figure 16.

It can be seen that the annual totals are all small, about 150 tonnes per year or less, compared to those in recent years which total around 1000 tonnes per year and have been up to about 2000 tonnes. The fishery in the Fish Board years was dominated by the Mackay Region, which contributed about 60% of the catch between 1969 and 1981. The Cairns Subregion contributed only about 25 tonnes per year, and the Townsville Subregion only about 15 tonnes per year.

Falls in harvest in 1967 and 1968 were probably associated with Tropical Cyclone Dinah in early 1967 which appears to have severely inhibited the growth of the fishery in the Mackay Region until 1969–70 (see Table 5, Figure 16 and Table 7).

The major fall in harvest in 1977, which is even more pronounced when the raw data for financial years are considered (Table 7), appears to have been associated with the introduction of a minimum legal size (MLS) in late 1976 (see Table 1), and, to a lesser extent, with Tropical Cyclone David which was a very large cyclone that struck the GBR in early 1976 and generated huge swells (see Table 5). Both the size limit and the cyclone may have had the effect of making fishers decide to target fish other than coral trout. The introduction of the MLS may have spurred fishers to travel further offshore: the catch reported from the Rockhampton station remained low in subsequent years.

Table 7: Queensland Fish Board annual returns, in tonnes, for coral trout, by reporting station. Coral trout was not reported prior to 1962–63. Minor stations are grouped with nearby major ones: Cairns+ includes Port Douglas; Home Hill+ includes Ayr; Bowen+ includes Proserpine; Rockhampton+ includes Yeppoon and Rosslyn Bay; Sunshine Coast comprises Tewantin, Mooloolaba and Caloundra; Metropolitan includes stations from Scarborough to the Gold Coast. Source: Database compiled by Halliday and Robins (2007).

Fiscal year	Cairns+	Innisfail	Townsville	Home Hill+	Bowen+	Mackay	Rockhampton+	Gladstone	Bundaberg	Maryborough	Tin Can Bay	Sunshine Coast	Metropolitan
1962–63	5.11	9.63	4.62	2.15	0.24	12.56	3.55	7.35	0.42	0.03	0.00	0.01	0.67
1963–64	4.65	8.91	4.23	1.36	0.19	8.39	1.99	9.42	0.37	0.03	0.02	0.07	0.00
1964–65	7.00	10.55	4.00	0.97	0.23	4.95	2.20	18.81	1.09	0.02	0.02	0.04	0.50
1965–66	9.44	16.37	13.38	0.77	0.34	6.51	1.58	14.77	6.00	0.06	0.02	0.04	3.27
1966–67	10.88	22.01	11.21	1.11	1.39	9.66	1.28	4.50	10.07	0.03	0.09	0.32	0.11
1967–68	9.61	19.04	6.89	2.03	1.26	10.81	1.28	2.85	5.29	0.00	0.07	0.02	0.03
1968–69	5.44	16.35	5.52	3.18	0.36	14.11	2.21	2.96	4.59	0.07	0.03	0.01	0.07
1969–70	24.65	10.16	8.07	4.19	0.00	73.76	12.49	7.21	13.07	0.09	0.03	0.37	0.92
1970–71	12.81	11.10	10.42	3.72	0.00	72.94	9.12	7.33	23.53	0.27	1.51	0.07	0.89
1971–72	13.15	14.26	11.74	5.33	0.27	72.91	13.96	6.50	24.53	0.02	0.19	0.11	0.57
1972–73	8.48	10.95	13.97	1.17	3.88	77.80	17.27	6.21	22.04	1.19	0.01	0.35	1.38
1973–74	8.08	5.56	13.15	0.84	2.34	87.98	11.15	7.55	17.11	0.79	0.56	0.60	1.59
1974–75	7.04	4.84	7.79	0.85	2.86	72.88	17.78	3.82	6.16	0.68	0.01	0.20	0.35
1975–76	10.17	8.65	5.89	0.90	4.31	63.00	20.93	6.35	10.98	0.22	0.04	0.22	0.35
1976–77	20.23	9.24	8.05	0.00	1.66	49.46	13.97	11.87	6.23	0.25	0.08	0.33	0.43
1977–78	18.27	12.67	8.24	0.00	0.09	2.06	10.06	7.83	4.41	0.03	0.01	0.25	0.40
1978–79	12.86	9.76	6.20	0.00	1.23	86.44	5.57	12.58	3.09	0.16	0.06	0.49	1.22
1979–80	10.83	8.51	6.62	0.05	13.98	103.27	3.61	19.94	9.46	0.19	0.13	0.42	1.44
1980–81	13.96	6.52	8.97	0.02	4.33	117.39	6.15	8.49	7.52	0.17	0.01	0.77	2.46

Table 8: Comparison of Queensland Fish Board 1979–80 commercial harvest size of all non-otter-trawl species from stations opposite the Great Barrier Reef to estimated catch values from the survey conducted by Hundloe (1985, pp. 65, 81). Catch value is from Hundloe; catch weight is from Fish Board records compiled by Halliday and Robins (2007). Values for Rockhampton are from 1978–79.

Hundloe region	**Catch value ($k)**	**Catch weight (t)**	**Av. price ($/kg)**
Cairns	1280.6	434.6	2.95
Townsville	1389.0	371.6	3.74
Mackay	1400.4	335.3	4.18
Rockhampton	1400.9	279.1	5.02
Total	**5470.9**	**1420.6**	**3.85**

The cause of the fall in harvest from 1973 to 1975 is unknown; it is unlikely that Tropical Cyclone Emily in early 1972 (see Table 5) would have been sufficient to cause it; Emily does not appear to have been an exceptionally large cyclone. Perhaps at that time the fishery was in a levelling-out phase after a period of rapid growth, and very easily accessible fishing locations were experiencing falls in abundance.

We note that there was relatively little impact of tropical cyclones on the GBR between 1981 and 1989, which may have allowed unhindered growth in commercial fishing effort during this time (see section 4.2.3 below).

4.2.2 Allocation of logbook harvest to six-nautical-mile grid squares

Logbook data from the CFISH database maintained by Fisheries Queensland began in 1988. For most years it records the catch location to six-nautical-mile resolution, which is roughly the scale of a reef and usually adequate to assign catches to Bioregions and Subregions. In the early years, however, location was often specified only to 30-nautical-mile resolution, which was not sufficient for allocation into Bioregions and Subregions.

Records that were specified only to a 30-nautical-mile grid square were split into six-nautical-mile grid squares following the split of catches in that 30-nautical-mile square (by all commercial vessels) that were recorded to six-nautical-mile resolution during the duration of the zoning (zoning dates are listed in Table 1, page 19). For example, if, between the two zoning dates 7 November 1983 and 3 April 1992 of the Cairns Section, the catches within a 30-mile grid square of this Section which were recorded to fine resolution were split 70% to six-mile square A, and 30% to six-mile square B, then the catches in that 30-mile square over the same time period that were not specified to six-mile resolution were allocated in the same 70-30 split.

The splitting of catches from 30-mile grid squares into six-mile grid squares was used only to find the annual harvest size in each Bioregion or Subregion. It was not used to analyse catch rates. The catch-rate analysis presented in section 4.6 used only daily catches that were specified to six-mile resolution.

4.2.3 Harvest sizes from logbook data

The annual harvests by Subregion inferred from logbook data are plotted in Figure 17. Comparing this to Figure 16, it appears that the fishery grew very rapidly between 1981 and 1991. It could be argued that the catches in Figure 16 are grossly understated due to marketing of product outside of the Queensland Fish Board, and that logbook totals between 1988 and 1991 are also underestimates due to incomplete take-up of the logbook system by fishers, but there is no strong evidence for this (*cf.* Table 8) and we prefer to conclude that there was indeed very sharp growth between 1981 and 1991. We remark also that this was the period during which wogging ceased to be a widely used method of commercial fishing (see section 1.3): possibly it was not feasible for these much larger catches to be taken by wogging, or there were insufficient numbers of fishers who possessed the specialist knowledge required for wogging.

No data on commercial harvest were available between 1981 (when the Queensland Fish Board ceased operations) and 1988 (when the commercial logbook system began), a period during which the apparent total harvest increased by about a factor of three. We assumed that the harvest increased exponentially from 1981 to 1988, i.e., at about 20% per year. Harvest from the Far Northern Region (Cape York, Lockhart River and Princess Charlotte Bay Subregions) was assumed to be negligible prior to 1988. This Region was much less accessible than the others.

It can be seen that the Mackay Region has made the dominant contribution to the overall coral trout harvest since 1969, followed by the Townsville Subregion, and the Swains and Cooktown Regions (Figure 16 and Figure 17).

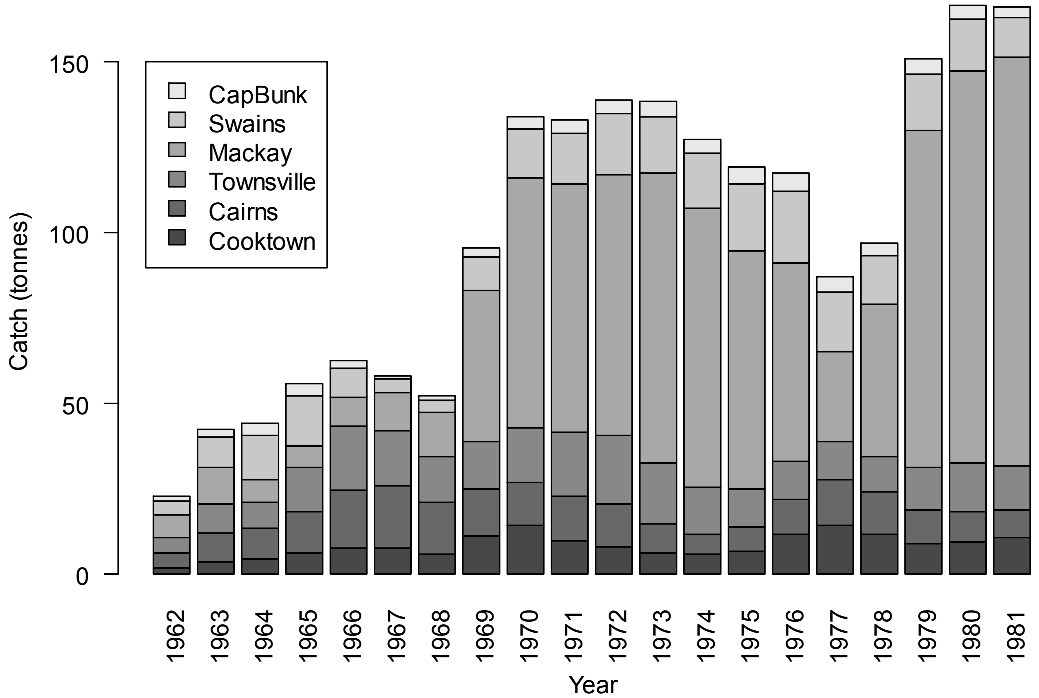


Figure 16: Early years of the commercial coral trout harvest by calendar year, assigned to Subregions. The final year, 1981, is actually the financial year 1980–81. The first year, 1962, is half of the financial year 1962–63. Source: Queensland Fish Board records compiled by Halliday and Robins (2007).

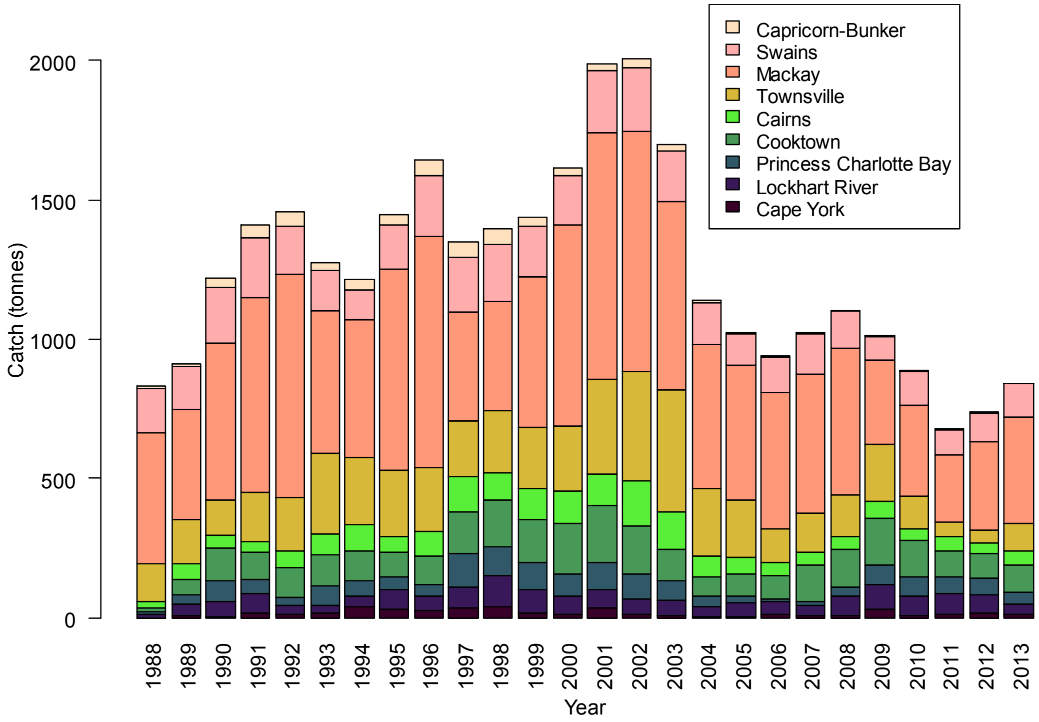


Figure 17: Annual commercial catch sizes, by Subregion ordered north (bottom of figure) to south (top of figure), from logbook data. The final year, 2013, is incomplete after October. Totals are slightly lower than Figure 9 because some records do not provide location. Source: CFISH logbook database maintained by Fisheries Queensland, February 2014.

The fall in total harvest between 2002 and 2006 came mainly from a decline in fishing effort associated with many factors:

- A major fishery restructure, including the introduction of harvest quotas, on 1 July 2004 (see section 1.3.2)
- Declaration of new green zones, also from 1 July 2004 (see section 1.3.2)
- High fuel prices
- Scarcity of skilled labour (dory operators), due to a mining boom in Australia
- Low product prices, due to epidemics of Severe Acute Respiratory Syndrome (SARS) and Avian Influenza (bird flu) in Asia.

Falls in harvest size associated with the following major tropical cyclone events are visible in Figure 17 (*cf.* Table 4):

- Cyclone Joy struck the northern GBR in late 1990.
- Cyclone Fran struck the Swains in early 1992.
- Cyclones Oliver, Roger and Rewa stuck the southern GBR from early 1993 to early 1994.
- Cyclone Justin stuck much of the GBR in early 1997. It had less effect on the Far Northern and Cooktown Regions, and the harvests there actually rose in 1997, probably due to some fishers temporarily relocating there.
- Cyclone Hamish stuck much of the GBR, but was most powerful in the south, in early 2009.
- Cyclone Yasi was also large enough to affect much of the GBR in early 2011, but was centred close to Townsville.

It should be noted that the fishery was in a strong growth phase up to 2001, which must have made falls in harvest during the period 1988–2001 appear less important than they actually were. Catch rates may show these falls better than catch sizes (see section 4.6 below).

4.3 Recreational catch size

4.3.1 Early surveys

The recreational survey by Hundloe (1985) estimated the total small-boat catch of fish (all species combined) off the coast opposite the Great Barrier Reef to be 6572 tonnes in the 1980 calendar year (see Table 9). "Small boats" were distinct from larger charter boats. This catch was 5.3 times the size of the commercial fishery in the Cairns Hundloe region, 5.1 times in Townsville, 3.3 times in Mackay, and 4.6 times in Rockhampton. The number of registered small boats in Queensland at the time was about 73,000, of which Hundloe estimated 14,887 to be fishing in the GBR region. The average catch calculated from these data is 441 kg per boat, which is high.

As remarked in the previous section, the commercial coral trout fishery was probably focussed more inshore in 1980 than it has been in recent years, especially until 1976 when the catch commercial harvest size had stabilised and the minimum legal size was introduced. The spatial distribution of commercial and recreational fishing effort on coral trout may have been very similar, and the above multiples of commercial catch may well apply to coral trout.

Table 9: Estimated recreational catch of fish (all species combined) harvested by small boats off the coast opposite the GBR, in calendar year 1980, reproduced from Hundloe (1985, p. 73). "Comm. mult." is the approximate multiple of the commercial harvest from Table 8.

Hundloe region	No. of fish	Av. weight (kg)	Harvest weight (t)	Comm. mult.
Cairns	864,300	2.67	2308.8	5.3
Townsville	818,900	2.30	1881.4	5.1
Mackay	497,800	2.22	1103.7	3.3
Rockhampton	951,400	1.34	1277.9	4.6
Total	**3,132,400**	**2.10**	**6571.8**	**4.6**

Hundloe (1985, p. 76) estimated the charter-boat harvest of fish (all species combined) in the GBR area at 700 tonnes in 1980, which was 0.5 times the commercial harvest in that year (*cf.* Table 8). He acknowledged that the charter boat industry consisted of distinct branches with very different fishing habits (some of which didn't fish at all), which made his estimate of catch size very uncertain. In common with small-boat recreational fishers, charter guests were allowed to sell some of their catch until 1990. Hundloe also estimates (p. 56–57) the numbers of boats and person-days of fishing by charter fishers, which are reproduced in Table 10 and used to split the charter catch into regions. Hundloe comments that charters from his Rockhampton region visited the Capricorn–Bunker and Swains reefs, but many trips in the Mackay region were non-fishing trips to the Whitsunday Islands.

In more recent years, harvest by charter boats is recorded as part of the commercial logbook database, compulsorily since 1996. Charter harvest is also counted in the diary surveys described in the next section, so the charter logbook catch does not need to be added in.

The data of Hundloe (1985) and the Queensland Fish Board records (section 4.2.1) allow the total recreational catch sizes of coral trout to be estimated (small-boat and charter fleets combined). The results of this are listed in Table 11.

A survey of participation in recreational fishing in Queensland in 1985 by the Australian Bureau of Statistics (ABS, 1986) found that about 572,000 Queensland residents aged 15 or over had engaged in recreational fishing in the year prior to the survey. This was a participation rate of 30% of the population. The number of boats involved State-wide was estimated at 85,000, which closely matches the figure given by Hundloe above, allowing for growth in the number of owners in the intervening five years.

Table 10: Estimated charter-boat fishing effort and harvest (all species combined) around the GBR in 1980 (plus or minus one year). Charter boats based in Cairns were almost exclusively deployed in the game fishery for species such black marlin (Istiompax indica) *and are not counted in the harvest. "Comm. mult." is the approximate multiple of the commercial harvest from Table 8.* Source: Hundloe (1985); regional harvests calculated from total harvest by assuming catch proportional to effort.

Hundloe region	Year	Boats	Effort (person days)	Harvest (t)	Comm. mult.
Cairns	1979	39	8,000	0	0.00
Townsville	1981	10	6,000	89	0.25
Mackay	1981	60	25,000	372	1.10
Rockhampton	1980	10	16,000	238	0.85
Total		**119**	**47,000**	**700**	**0.50**

Table 11: Total recreational catch (small-boat and charter-boat combined) of coral trout (several species combined) by weight in 1980, estimated from the commercial catch in that year (Figure 16) and the commercial catch multiples listed in Table 9 and Table 10. The multiples from the Cairns Hundloe region were applied to the Cooktown and Cairns stock assessment Subregions, and the multiples from the Rockhampton Hundloe region were applied to the Swains and Capricorn–Bunker Subregions.

Subregion	Comm. mult.	Harvest weight (t)
Cooktown	5.30	49
Cairns	5.30	46
Townsville	5.35	76
Mackay	4.40	506
Swains	5.45	83
Capricorn–Bunker	5.45	21
Total	**4.70**	**782**

The ABS survey defined four fishing areas that covered the Great Barrier Reef: Area 18 (equivalent to the Far Northern Region used in this assessment), Area 15 (Cooktown–Cairns–Townsville), Area 13 (Mackay and the Swains), and Area 11 (Capricorn–Bunker). The estimates of the number of fishers that fished these areas were 1000, 12,000, 11,000 and 10,000 respectively. This provides a reasonable match to Hundloe's figure of 14,887 boats (about two fishers per boat).

Another ABS survey of home food production (ABS, 1994) estimates that 245,900 Queensland households fished in 1991 (actually the year ending April 1992), and caught 7284 tonnes of seafood for home consumption, 5732 t of which was fish. The total seafood catch was partitioned into 2516 t from Brisbane and 4768 t from the rest of Queensland. Assuming that the ratio of fish to other seafood was the same in Brisbane and the rest of Queensland, this gives an estimate of 3752 t of fish from the rest of Queensland, which is a fall of 42.91% from Hundloe's estimate from 1980.

These survey estimates imply that about half the recreational catch in 1980 was sold and not consumed at the home of the fisher. This practice was permitted until 1990 (see Table 1, page 19). In recent years, commercial fishers have stated their belief that this semi-commercial part of the recreational fishery was large in the 1980s. It also explains how the average annual catch per small boat could have been so high in Hundloe's survey.

Higgs (1995, Figure 9) shows that the proportion of coral trout by number in the catch by fishing clubs in the Townsville region (Hundloe region, as in Table 9) remained fairly constant around 37% from 1961 to 1989, after which the clubs indulged in more night-time fishing targeting fish from the family Lutjanidae (tropical snappers). He also comments that the number of privately registered boats in Queensland increased five-fold from 1968 to 1994, and quotes a report by Blamey and Hundloe (1993) as estimating the number of boats in waters off the coast opposite the GBR as 24,300, which caught between 3500 and 4300 t of fish.

For the other Hundloe regions, the overall fishing-club catch percentages of coral trout in the catches are (Higgs, 1995, Figure 6) about 18% for Cairns, 45% for Mackay and 14% for Rockhampton, which make it clear that coral trout was an attractive target species for recreational fishers, especially in the Mackay region. We note that the figure of 45% coral trout for Mackay agrees very closely with the 1980 harvest estimates of 506 t of coral trout (which may be viewed as very large) (Table 11) and 1104 t of fish of all species (Table 9).

Finally, Higgs states that the average weight of fish in the recreational fishery for coral trout and red throat emperor off Townsville fell sharply from the early 1960s to the late 1960s, but fell only slightly after that. This implies that the early 1960s were probably formative years for the fishery, and that the recreational coral trout fishery developed about the same time as the commercial fishery.

The catch-size estimate from Blamey and Hundloe quoted by Higgs confirms the ABS estimate. If the Hundloe (1985) estimate for 1980 is correct, this confirms that the 1990 prohibition of the sale of fish by recreational fishers probably had a big effect on the fishery, cutting the recreational harvest by about half.

4.3.2 NRIFS and SWRFS diary survey results

Recreational catches of fish in Queensland have been measured by State-wide diary surveys since 1997:

- Surveys conducted by Fisheries Queensland, known as RFISH, in 1997, 1999, 2002 and 2005 (Higgs, 1999; 2001; Higgs et al., 2007; McInnes, 2008).
- An Australian national survey (the National Recreational and Indigenous Fishing Survey, NRIFS) was conducted in 2000 (actually May 2000 to April 2001) and used different methodology. It was funded by the Australian Government's Fisheries

Research and Development Corporation (FRDC, project number 99/158) (Henry and Lyle, 2003).

- The NRIFS methodology was adopted by Fisheries Queensland for the State-wide survey in 2011, known as SWRFS (State-Wide Recreational Fishing Survey, pronounced “Swirfs”) (Taylor et al., 2012).

All of the above surveys consisted of two stages: a preliminary telephone survey to measure the participation rate of residents in each statistical area, followed by a year-long diary survey of telephone respondents who participated in recreational fishing. The final results were weighted up by the locations where the fishers lived, not where they fished: counts of fish caught were scaled up by the number of people living in each statistical area. This survey design was not biased, but tended to sample more residents of the densely populated south-eastern part of Queensland. Greater uncertainty is associated with catches by residents of north Queensland.

Inspection of the data indicated that, in general and over all species, the RFISH published estimates of recreational catch sizes were consistently larger than the NRIFS-SWRFS ones. Recreational fishing specialists in Fisheries Queensland have ascribed this to differences in survey methodology. The NRIFS and SWRFS surveys made every effort to avoid drop-out of participants during the survey, and involved the use of carefully prepared prompt sheets by telephone interviewers who followed up survey participants and suggested various types of fish that they might have caught. They also explicitly took account of the number of fishers in a party, using it to divide the total number of fish caught by the party in order to calculate the diary fisher’s share.

RFISH participation was subject to a greater dropout rate than in the NRIFS-SWRFS surveys, and the assumption had to be made that participants who dropped out caught as many fish on average as the participants who remained until the end of the survey. In practice, participants are likely to have dropped out not merely from the survey but also from fishing, and therefore to have caught fewer fish than the participants that remained. If this occurred, the RFISH results would be overestimates. The RFISH methodology was also less clear in its handling of catches that were shared between members of a fishing party, only one of whom would have been a diary respondent. We do not intend these points as criticism of the RFISH program, but an acknowledgement that methodology for such surveys was refined over time.

We have followed the recommendation of Fisheries Queensland in regarding the NRIFS-SWRFS estimates as the best available, and the RFISH ones as potential overestimates.

Fishing locations were specified precisely in the SWRFS survey, but in the NRIFS survey they were specified only in broad regions. Therefore it was easy to split the SWRFS results into the Subregions used in the stock assessment. The SWRFS results are listed in Table 12. Inner-shelf fishing locations were defined to be those in the GMRMPA Bioregions (see Figure 6) that were close to shore and not frequented by commercial fishers in the logbook data; these were locations beginning with “RE”, “RF” or “RHC”, plus the part of Bioregion “RD” that was in the Princess Charlotte Bay Subregion.

The NRIFS results are listed in Table 13, categorised by the NRIFS regions which are listed in Table 14. They come from reanalysis of the raw data of Henry and Lyle (2003). We weighted the diary catches by the field `PersonDiaryWeight` in the raw data. Our calculated total catch is a few per cent lower than that reported by Henry and Lyle; the reason for this is unknown, but the difference is much less than one standard error. We calculated standard errors using household and fishing trip as the experimental units and a Poisson generalised linear model (GLM) for the number of fish caught, with over-dispersion. The dispersion parameter, from which standard errors were calculated, was estimated by the mean deviance. Our standard error for the total Queensland catch of coral trout is roughly twice the published value from Henry and Lyle.

It is evident that the classification of inshore versus offshore fishing locations in the NRIFS bears little resemblance to the inner-shelf versus offshore separation used above. "Inshore" fishing in the NRIFS denoted fishing within 5 km of the coast (Henry and Lyle, 2003, p. 113), which in most Regions is much less than the distance used in the stock assessment. Therefore the inshore versus offshore split in the NRIFS data was not used. NRIFS region 7 was anomalous with a high catch recorded as inshore, although this was subject to a very high standard error. The reefs in the GBR come very close to the shore in this region, and it is conceivable that some reefs classified as inshore by fishers should be classified as offshore for stock assessment purposes. In common with the other regions, we did not make any use of the inshore versus offshore split in NRIFS region 7.

Table 14 lists the proportions in which NRIFS catches were allocated to the stock assessment's Subregions; these proportions were informed by the SWRFS results (Table 12).

The resulting NRIFS catches by Subregion are listed in Tables 15 and 16: Table 15 includes fishers who live anywhere in Australia, whereas Table 16 includes only Queensland residents, which makes it directly comparable to Table 12.

Finally, Table 17 lists the catch estimates from the SWRFS survey, scaled up to cover both Queensland residents and interstate visitors, using data from the NRIFS survey.

Table 12: Results from the 2011 SWRFS diary survey, providing numbers of coral trout kept, released and total, with standard errors, by Subregion. Suffices "I" and "O" stand for inner-shelf and offshore. Source: Raw data from Taylor et al. (2012).

Subregion	**Kept**	**s.e.**	**Released**	**s.e.**	**Total**	**s.e.**
Cape York	0	–	0	–	0	–
Lockhart River I	0	–	0	–	0	–
Lockhart River O	2650	3176	2319	2969	4969	5725
Princess Charlotte Bay I	0	–	0	–	0	–
Princess Charlotte Bay O	1067	1305	1992	2551	3059	3497
Cooktown I	0	–	0	–	0	–
Cooktown O	3187	2269	2305	2212	5492	3811
Cairns I	13013	6643	7130	5566	20143	10511
Cairns O	9352	4419	5365	3938	14717	7110
Townsville I	12084	4700	5809	3435	17893	6812
Townsville O	12051	5739	5836	3082	17887	7951
Mackay I	9907	4012	10990	4735	20898	7001
Mackay O	3588	2504	2038	1697	5626	3446
Swains	14285	8026	13644	9937	27930	14483
Capricorn–Bunker I	12636	7846	10049	10313	22685	17161
Capricorn–Bunker O	3123	1911	1130	1342	4253	2596
South of GBR	3821	2741	3948	3742	7768	4969
Gulf of Carpentaria	2730	2180	288	389	3018	2440
Total GBR inner-shelf	**48718**	**12409**	**35469**	**13272**	**84187**	**22941**
Total GBR offshore	**49303**	**12573**	**34630**	**12412**	**83933**	**20988**
Total GBR	**98021**	**18124**	**70099**	**18425**	**168120**	**31792**

Table 13: Results from the 2000 NRIFS diary survey, providing numbers of coral trout kept, released and total, with standard errors, by NRIFS fishing region number from Table 14. The numbers include coral trout caught in Queensland by fishers living anywhere in Australia. Source: Reanalysis of raw data from Henry and Lyle (2003).

NRIFS fishing region	Kept	s.e.	Released	s.e.	Total	s.e.
7 inshore	12046	12324	760	977	12806	12608
7 offshore	2138	1823	453	822	2591	2257
6 inshore	2801	2070	2177	1910	4978	3068
6 offshore	63690	17798	41021	15667	104711	30536
5 inshore	4257	2944	7012	4262	11269	6264
5 offshore	51342	19417	25224	9054	76566	25618
4 inshore	1897	1365	1934	1511	3831	2275
4 offshore	57891	20528	43915	20730	101806	35360
3 inshore	3991	3034	796	1444	4788	3811
3 offshore	53103	22162	11201	8027	64305	25883
South of GBR	2401	2312	2197	2686	4598	3839
Gulf of Carpentaria	4953	7822	4219	8102	9171	13068
Torres Strait	213	483	0	0	213	532
Inland	1606	2796	0	0	1606	3072
Total GBR inshore	**24992**	**13298**	**12679**	**5202**	**37671**	**15115**
Total GBR offshore	**228163**	**43136**	**121815**	**28779**	**349978**	**61769**
Total GBR	**253155**	**45842**	**134494**	**29840**	**387649**	**64871**

Table 14: Fishing regions used for recording in the 2000 NRIFS diary survey, and the proportions in which catches were assigned to the model's Subregions. The proportions are averages of the 1999 and 2002 RFISH surveys (Table 18), except that the Cape York, Lockhart River and Princess Charlotte Bay Subregions have negligible RFISH catches and are assigned zero. Inland fishing regions (numbers 12–17 and 19), which consisted of estuaries, rivers and dams, are omitted. Suffices "I" and "O" denote inner-shelf versus offshore.

NRIFS fishing region		Subregion	Proportion
1	NSW border to Double Island Point	South of GBR	1.00
2	Double Island Point to Round Hill Head	South of GBR	1.00
3	Round Hill Head to Cape Clinton	Capricorn–Bunker I	0.20
		Capricorn–Bunker O	0.33
		Swains	0.47
4	Cape Clinton to Gloucester Head	Mackay I	0.72
	(Gloucester Island)	Mackay O	0.28
5	Gloucester Head to Lucinda Point	Townsville I	0.25
		Townsville O	0.75
6	Lucinda Point to Cape Tribulation	Cairns I	0.14
		Cairns O	0.86
7	Cape Tribulation to Tern Cliffs	Cooktown I	0.17
		Cooktown O	0.83
8	North of Cape York	Torres Strait	1.00
9	Cape York to Thud Point (S of Weipa)	Gulf of Carpentaria	1.00
10	Thud Point to 140° E	Gulf of Carpentaria	1.00
11	140° E to NT border	Gulf of Carpentaria	1.00
18	Moreton Bay	South of GBR	1.00

Table 15: Results from the 2000 NRIFS diary survey, providing numbers of coral trout kept, released and total, by Subregion as used in the stock assessment, taken by fishers living anywhere in Australia. Harvest = kept number multiplied by 1.576 kg (see section 1.5.3). Source: Raw data from Henry and Lyle (2003), allocated according to *Table 14*.

Subregion	**Kept**	**Released**	**Total**	**Harvest (t)**
Cape York	0	0	0	0.00
Lockhart River	0	0	0	0.00
Princess Charlotte Bay	0	0	0	0.00
Cooktown inner-shelf	2411	206	2617	3.80
Cooktown offshore	11772	1007	12779	18.55
Cairns inner-shelf	9309	6048	15356	14.67
Cairns offshore	57182	37150	94332	90.12
Townsville inner-shelf	13900	8059	21959	21.91
Townsville offshore	41699	24177	65876	65.72
Mackay inner-shelf	43047	33011	76059	67.84
Mackay offshore	16741	12838	29578	26.38
Swains	26834	5639	32473	42.29
Capricorn–Bunker inner-shelf	11419	2400	13818	18.00
Capricorn–Bunker offshore	18841	3959	22800	29.69
Total GBR inner-shelf	**80086**	**49724**	**129810**	**126.22**
Total GBR offshore	**173069**	**84770**	**257840**	**272.76**
Total GBR	**253155**	**134494**	**387649**	**398.97**

Table 16: Results from the 2000 NRIFS diary survey by Subregion, counting only coral trout caught by Queensland residents, for comparison to Table 12. Source: Raw data from Henry and Lyle (2003), allocated according to Table 14.

Subregion	**Kept**	**Released**	**Total**
Cape York	0	0	0
Lockhart River	0	0	0
Princess Charlotte Bay	0	0	0
Cooktown inner-shelf	2291	128	2419
Cooktown offshore	11187	625	11812
Cairns inner-shelf	8984	5916	14900
Cairns offshore	55188	36342	91530
Townsville inner-shelf	13547	7564	21110
Townsville offshore	40640	22691	63331
Mackay inner-shelf	21415	22097	43512
Mackay offshore	8328	8593	16921
Swains	17926	5265	23191
Capricorn–Bunker inner-shelf	7628	2240	9868
Capricorn–Bunker offshore	12586	3696	16283

Table 17: Results within the GBR from the 2011 SWRFS diary survey, providing numbers of coral trout kept, released and total, by Subregion, scaled up using data from the NRIFS survey to account for interstate visitors. Harvest = kept number multiplied by 1.576 kg (see section 1.5.3); I denotes inner-shelf, O denotes offshore.

Subregion	**Kept**	**Released**	**Total**	**Harvest (t)**
Cape York	0	0	0	0.00
Lockhart River I	0	0	0	0.00
Lockhart River O	2650	2319	4969	4.18
Princess Charlotte Bay I	0	0	0	0.00

Princess Charlotte Bay O	1067	1992	3059	1.68
Cooktown I	1135	2400	3535	1.79
Cooktown O	3353	3712	7066	5.28
Cairns I	13483	7289	20772	21.25
Cairns O	9690	5484	15174	15.27
Townsville I	12398	6190	18588	19.54
Townsville O	12365	6219	18583	19.49
Mackay I	19915	16419	36334	31.39
Mackay O	7212	3044	10256	11.37
Swains	21385	14615	35999	33.70
Capricorn–Bunker I	18915	10764	29679	29.81
Capricorn–Bunker O	4674	1210	5885	7.37
Total GBR inner-shelf	**65846**	**43061**	**108908**	**103.77**
Total GBR offshore	**62396**	**38596**	**100992**	**98.34**
Total GBR	**128243**	**81657**	**209900**	**202.11**

4.3.3 RFISH diary survey results

The RFISH catches are summarised in Table 18. Data are weighted by the fields `w_kept` (1997, 1999 and 2002) and `WeightedKept` (2005), with no extra adjustment for dropout of participants. The results will be discussed in the next section, along with the NRIFS-SWRFS results. The latter were informed by the regional splits of the 1999 and 2002 RFISH surveys.

Table 18: Estimated numbers of retained coral trout (several species combined) from RFISH surveys, by Subregion and year of survey. Only catches for which the location was adequately specified are included. The numbers are not directly comparable to the NRIFS-SWIRFS results, due to differences in methodology, but can be used to gauge trends. "I" denotes inner-shelf, and "O" offshore. Source: Reanalysis of raw data from RFISH surveys.

	Year			
Subregion	**1997**	**1999**	**2002**	**2005**
Cape York	0	0	0	0
Lockhart River I	252	0	0	0
Lockhart River O	548	0	0	0
Princess Charlotte Bay I	686	0	216	764
Princess Charlotte Bay O	0	0	0	0
Cooktown I	761	4031	1849	645
Cooktown O	17449	15276	13112	1974
Cairns I	9832	6721	7062	3809
Cairns O	45171	39905	45801	14727
Townsville I	5503	7321	9655	6738
Townsville O	19713	32911	17109	27337
Mackay I	20625	30435	28121	33547
Mackay O	4625	4766	17951	2658
Swains	6720	13500	12654	5786
Capricorn–Bunker I	4168	5942	5024	7892
Capricorn–Bunker O	5522	6585	11350	9226
South of GBR	3366	8577	2133	2548
Gulf of Carpentaria	951	2979	789	586
Torres Strait	950	1877	2661	308
Total GBR inshore	**41827**	**54450**	**51927**	**53395**
Total GBR offshore	**99748**	**112943**	**117977**	**61708**
Total GBR	**141575**	**167393**	**169904**	**115103**

4.3.4 Discussion of the diary surveys

Comparison of Table 17 to Table 15 shows dramatic falls in the catches between 2000 and 2011, especially in offshore locations. Table 18 shows that the falls offshore were particularly sharp between 2002 and 2005. The falls may have had the following causes:

- Decline in the recreational fishing participation rate: for Queensland residents this fell steadily from 30% in 1985 to 23% in 2000 and 17% in 2010, and the total number of boat-days that they fished in Queensland fell from 1.7 million in 2000 to 1.2 million in 2010 (ABS, 1986; Henry and Lyle, 2003; Taylor et al., 2012).
- High fuel prices could have discouraged fishers from travelling far offshore.
- New green zones declared in 2004 could have discouraged fishers.
- The 2011 survey was undertaken in the year of Cyclone Yasi and two years after Cyclone Hamish, when catch rates of coral trout were low. Yasi particularly affected the Cairns and Townsville Subregions.

The fall in the Cooktown Region offshore was extremely sharp, and is shown by both the NRIFS-SWRFS surveys and the RFISH surveys. The far northern Subregions from Cape York to Cooktown currently have very low levels of recreational fishing.

Table 18 shows low catches in all Subregions from Townsville south in 1997, due to Cyclone Justin.

Between 1980 and 2000, growth in human populations in the cities of Cairns and Townsville was associated with rises in the recreational harvest of coral trout in those Subregions (Table 11 and Table 15), from 46 t to 105 t and 76 t to 88 t respectively. The Capricorn–Bunker Region also rose, from 21 t to 48 t. The Mackay Region, on the other hand, experienced a big fall in recreational harvest, from an estimated 506 t to 94 t, which may have been caused largely by the prohibition of unlicensed, quasi-commercial fishing in 1990 (Table 1, page 19).

All Subregions experienced falls in recreational harvest of coral trout between 2000 and 2011 (Table 15 and Table 17). There was, however, an increased preference for inner-shelf locations which produced rises in the inner-shelf harvest in the Cairns and Capricorn–Bunker Subregions.

Interstate visitors, in 2000 at least, overwhelmingly chose to fish in the Mackay, Swains and Capricorn–Bunker Regions, and took minimal catches between Cape York and Townsville (NRIFS raw data).

Some results in Table 18 appear anomalous, but this may be due to short-term changes in the habits of recreational fishers. For example, a large amount of effort appears to have moved from Townsville Subregion offshore to Mackay offshore in 2002, but then moved back again to Townsville in 2005. Many recreational fishers may simply fish where others fish, rather than make their own decisions about the where the fish are, meaning that much of the community may change fishing locations according to influential fishers and fishing reports.

4.4 Indigenous harvest size

The indigenous harvest of coral trout (several species combined) in north Queensland was estimated at 7004 fish in 2000 (Henry and Lyle, 2003, p. 121). This comprised most of the indigenous harvest from across northern Australia. Only a minority of the Australian harvest (44.15%) was taken more than 5 km from the coast (Henry and Lyle, 2003, p. 122). Probably only a very small percentage would have been taken in locations that were designated “offshore” for stock assessment. The indigenous communities sampled by Henry and Lyle range from Cairns north to Cape York and around into the Gulf of Carpentaria. Coral trout are present in the Gulf of Carpentaria but are much less commonly caught than on the Queensland east coast (see Table 12, Table 13 and Table 18). Henry and Lyle (p. 102) identify eight coastal communities on the tip of Cape York who would have done much of their fishing in Torres Strait, three around Lockhart River, seven around Cooktown, and one near Cairns. Most of the communities are small (around 250 people), but the one near Cairns (Yarrabah) had about 2000 inhabitants.

The indigenous catch was distributed between the Lockhart River, Cooktown and Cairns Subregions in proportion to the estimated human population sizes in these locations. It was all assumed to come from inner-shelf locations, and therefore ended up being excluded from the stock assessment which used only offshore locations.

4.5 Reconstruction of harvest size history

For input to the stock assessment's population dynamic model, the time series of commercial and recreational coral trout harvest (several species combined) were reconstructed from the data described above. The stock assessment includes only offshore locations, on the grounds that, at least from 1988, inner-shelf locations were not frequented by commercial coral trout fishers and generally had substantial numbers of *Plectropomus maculatus*, often exceeding the numbers of *P. leopardus* (see section 4.1).

The following assumptions underlay the reconstruction:

- Both the commercial and recreational fisheries started to become significant in 1953.
- Commercial harvest sizes increased linearly over ten years, from zero in 1952 to the 1962–63 Queensland Fish Board values in calendar year 1962 (Table 7, Figure 16).
- Until 1981, the proportions of commercial catch that were taken in inner-shelf locations in Subregions other than Mackay were the same as for the recreational fishery in 1997 (Table 18). In Mackay this assumption was incompatible with the early years of logbook data, and therefore the inner-shelf catch was capped at the 1968 level, which was when the fishery in Mackay began to expand very strongly.
- Until 1980, recreational harvests were proportional to commercial harvests, the constants of proportionality in the different Subregions being given by Table 11. The inner-shelf proportions of recreational catch were as in Table 18 in all Subregions, including Mackay.
- In the Cape York, Lockhart River, Princess Charlotte Bay and Cooktown Subregions, the split of commercial harvest between Subregions was the same as in the logbook data for 1989. The 1988 logbook data (first year of logbook data) were unusable in these Subregions, as they were much lower than 1989 and it was clear that most of the catch had not been recorded.
- Recreational harvests in the Cape York, Lockhart River and Princess Charlotte Bay Subregions were negligible prior to 2000. From 2000 onwards, the offshore recreational harvests in these Regions were equal to the 2011 SWRFS estimates, and the 2000 offshore harvest in the Cooktown Region was correspondingly less than that in Table 16.
- The commercial harvest increased geometrically by a fixed percentage (about 20%) per year in each offshore Subregion from 1981 (Table 7, Figure 16) to 1988 (Figure 17) (1989 for the four northernmost Subregions). This filled in the gap between the end of the Queensland Fish Board records and the beginning of the logbook database.
- From 1988 onwards (1989 for the four northernmost Subregions), the commercial harvest was equal to the recorded logbook harvest.
- Recreational harvest followed an underlying geometric trend, changing by a fixed percentage each year in each offshore Subregion, from 1980 (Table 11) to 2000 (Table 15), but this was overlaid by a fall of 21.45% in 1990 and a further identical fall in 1991, to account for the effect of prohibition on recreational fishers selling their catch from mid-1990 (see section 4.3.1).
- In 1997 and 1998, the recreational harvest in each Subregion was a fixed percentage higher or lower than expected, caused by Cyclone Justin. The ratio of the 1997 harvest (inner-shelf and offshore combined) to the 1999 harvest in a particular

Subregion was set equal to the ratio of the 1997 RFISH estimate to the 1999 RFISH estimate (Table 18).

- Recreational harvest was constant from 2000 to 2002, and then followed a geometric trend, changing by a fixed percentage each year in each Subregion, from 2002 to 2011. We did not attempt to include the short-term variation in locations fished by recreational fishers that is indicated by large changes (e.g., factors of two or more) in the RFISH data in particular regions.
- Recreational harvest was constant from 2011 to 2013.

The harvest reconstructions are shown graphically by Subregion in Figure 18, and by sector in Figure 19.

The harvest in inner-shelf locations (comprising mainly the Bioregions beginning with “RE” or “RHC” in Figure 6, page 14) is mainly recreational and appears to have decreased dramatically since 1980, while the offshore harvest is mainly commercial and increased greatly from 1980 to 2002. The inner-shelf harvest is dominated by the Mackay Region with its islands, including the world-famous Whitsunday Islands in the north, which are part of the greater Cumberland Group, the inshore Newry Group, and the extensive Northumberland Group in the south. The island and other inner-shelf locations are favoured by recreational fishers probably for their natural beauty and ease of access, while commercial fishers favour offshore locations, perhaps mainly for the greater certainty of species that they will catch there. Segregation between the commercial and recreational sectors must reduce conflicts and improve the quality of fishing experience for participants in both sectors.

The dominance of the Mackay Region since the early 1980s lends credence to statements by fishers that commercial fishers from Mackay came to fish in the Swains after 1984 due to falling catch rates closer to home (see Chapter 1).

Many effects of tropical cyclones are apparent in the reconstructions. We have shown the cyclones that are known to have been exceptionally large in size, or to have been powerful and have moved directly along the Great Barrier Reef (e.g., Hamish). Cases could be made for other cyclones such as Emily (1972), Simon (1980) and Freda (1981) (see Table 5), but more research needs to be done before they could be justified as having big and long-lasting effects on the GBR. As might be expected, it can be seen that cyclones have a bigger effect offshore than they do in inner-shelf locations, as they usually dissipate quickly after coming into contact with land.

The event with the biggest impact of all was the fishery restructure in July 2004, which seems to have caused major falls in harvest from 2002 to 2006. The major elements of this restructure were the introduction of commercial catch quotas as part of the management of the fishery, and the coincidental declaration of new green zones (Marine Protected Areas) by the Great Barrier Reef Marine Park Authority at exactly the same time (see section 1.3.2 in Chapter 1). Another big impact of fishery management appears to have come from the introduction of the minimum legal size of 35 cm in December 1976 (see Table 1). This may have actually spurred the expansion of the commercial fishery as fishers quickly learnt to live with it and travel further afield.

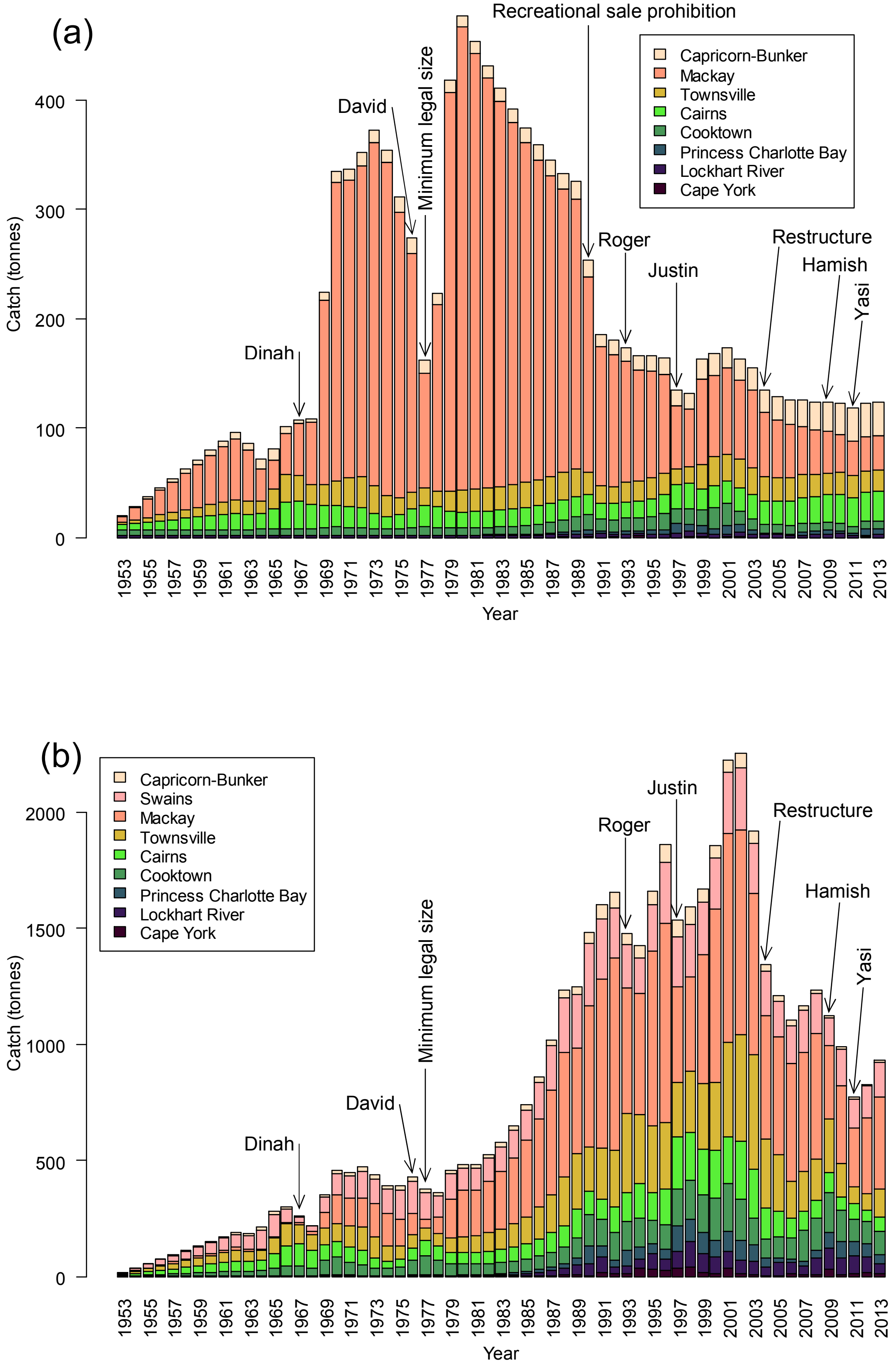


Figure 18: Reconstructed catch histories of coral trout (several species combined) by Subregion for (a) inner-shelf and (b) offshore fishing locations, also showing major tropical cyclones and fishery management events. The final year, 2013, is incomplete after October.

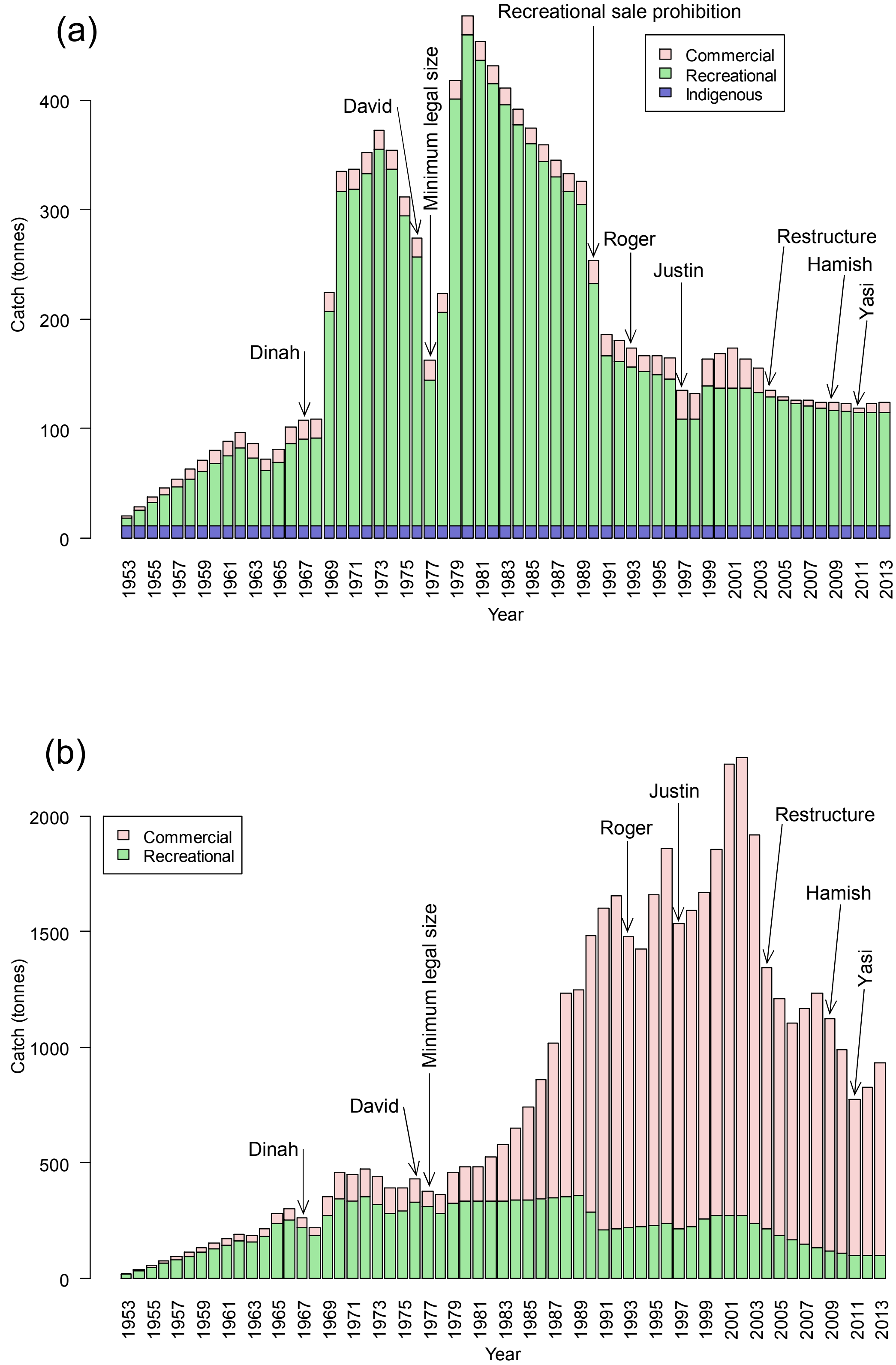


Figure 19: Reconstructed catch histories of coral trout (several species combined) by sector for (a) inner-shelf and (b) offshore locations, also showing major tropical cyclones and fishery management events. The final year, 2013, is incomplete after October.

4.6 Analysis of commercial catch rates

4.6.1 Methods

Commercial fishery logbook data were analysed to provide a time series of standardised catch rates of coral trout from the beginning of commercial logbook records. Logbook records actually began in 1988, but most records for the first few years specified the location to only thirty-nautical-mile resolution, not six-nautical-mile. The finer resolution was desired for the catch rate analysis, in order to distinguish Bioregions. Therefore the logbook data were deemed unsuitable for catch-rate analysis until mid-1991.

The analysis took the form of a generalised linear model (GLM) (McCullagh and Nelder, 1989) with quasi-Poisson error distribution. Whereas an ordinary linear model, given data y_j, would calculate the fitted daily catches $\hat{\mu}_j$ by minimising the sum of squares

$$\sum_j \left(y_j - \hat{\mu}_j\right)^2,$$

the quasi-Poisson model instead minimises the *deviance*

$$2\sum_j \left\{\hat{\mu}_j - y_j + y_j \log\left(y_j / \hat{\mu}_j\right)\right\}.$$

The prefix "quasi" means that the data are not treated as integers, as they would be under a strict Poisson model. The data in the application here are daily catches, measured in kilograms, and are mostly not integers. Many sources, such as McCullagh and Nelder (1989), call this simply a Poisson model. We have followed the statistical software *R* (R Core Team, 2013), which we used for the analysis, in favouring the term "quasi-Poisson" to emphasise that the model is more abstract: catches are not counts of units of one kilogram.

In the strict Poisson model the data are counts of independent events (e.g., individual fish which do not school), which follow a Poisson distribution; the probability function for this distribution is

$$p(y) = e^{-\mu} \mu^y / y!$$

for $y = 0, 1, 2, \ldots$. In this case the dispersion parameter is assigned the value 1. The Poisson model can be extended to allow for "over-dispersion", in which the events are no longer independent but occur in clumps (e.g., schools of fish). Then the counts no longer follow a Poisson distribution. The dispersion parameter takes a value roughly equal to the average size of a clump (average number of fish in a school). Further theoretical extension from the Poisson to the quasi-Poisson model allows non-integer data that are not counts; then no such distinction between ideally dispersed and over-dispersed data is possible.

The major advantage of the quasi-Poisson model for catch rates is that it automatically weights the data correctly. For example, if one vessel fishes for one day with two dories, the standard deviation of its catch will be $\sqrt{2}$ times that of another boat that fishes with only one dory, if all other things are equal; the quasi-Poisson model will produce the same results whether the experimental unit is a dory day or a primary-vessel day (provided that the number of dories is known and is accounted for in the model). The same applies if for some unrecorded reason one vessel is twice as efficient as another (perhaps because the skipper and dory operators are more skilful). The model estimates an efficiency parameter for each vessel, and automatically downweights the vessels with lower efficiency. This property is not possessed by other models such as the ordinary linear model (in which the ratio of standard deviations would be 1 instead of $\sqrt{2}$, thereby overweighting the more productive vessels) or the loglinear model or gamma model (both of which would have a ratio of 2 instead of $\sqrt{2}$, thereby underweighting the more productive vessels).

The model analysed daily catch records by primary vessel. The logbook data were first processed to produce a single record for each vessel–day combination on which any catch of *Plectropomus* or *Lethrinus* species was recorded. A vessel identifier was defined as the combination of the two fields `AuthorityChainNumber` and `Boatmark` from the CFISH database. A data record was then specified by the combination of vessel identifier and date. Where a vessel fished two different locations on the same day, all the catch was assigned to the square with the highest catch. All records that had a stated duration of more than one day were excluded, as were all records not specified to six-nautical-mile resolution, all records in which number of dories was not specified and could not be inferred, and all records for vessels that fished in only one calendar year. The last exclusion was done for the reason that if a vessel fishes in only one year and the model has to estimate an efficiency factor for it, this leaves no information that would allow that vessel to contribute to the annual time series of catch rates. The final number of data records analysed was 268,527, reduced from a total number of 497,563. The majority of exclusions were due to lack of six-nautical-mile resolution.

The catch rate analysis was programmed in the statistical software *R* (R Core Team, 2013). It was not possible to use the function `glm` to fit the generalised linear model because the model contained a large number of parameters and would have needed far more computer memory that was available. Instead, the GLM was programmed as a relatively simple, slowly converging algorithm that did not require a large amount of memory.

The parameters used in the GLM are listed in Table 19. The total number of parameters fitted was 2878. Locations between which coral trout abundance might differ were specified as "Subbioregions"; each Subbioregion was the intersection of a Bioregion with a Subregion. The population dynamic model attempted to follow the Bioregions defined by GBRMPA's expert committees, but some of them had to be split due to north–south changes in fishing intensity. Each Subbioregion belonged to a unique Subregion, over which catch rates were defined. The regional structure is described in more detail in section 6.2 in Chapter 6 (page 84).

The large number of parameters was desired in order to model effects such as tropical cyclones, which can hit some Subregions but not others, and can have a sudden effect in a particular month of a particular year. Such effects could not be represented by two-way interactions of Year, Month (1–12) and Subregion, but needed the full three-way interaction.

Most of the parameters had multiplicative effects. Taking the vessel effect as an example, a multiplicative effect means that if vessel A is twice as efficient as vessel B, then it will catch twice as many fish per day as vessel B at a given time and place, whatever that time and place may be. If, for example, the abundance of fish decreases at some time, then catches from both vessels will both be reduced and vessel A will still catch twice as many fish as vessel B.

The effects of number of dories and number of crew were unusual in that they were additive. Extra dories do not increase the catch exponentially; for example, if the catch from two dories is double that from one dory, we do not expect the catch from three dories to be double that from two dories, as would be implied by a multiplicative model. Rather, we expect each extra dory to increase the catch by the same amount, according to an additive model.

The final model was given by the equation

$$\hat{\mu}_j = \left\{1+\beta^{(1)}(d_j-4)+\beta^{(2)}I(d_j=0)+\beta^{(3)}x_j\right\} \\ \times \exp\left\{\beta^{\text{vessel}}_{v(j)}+\beta^{\text{Subbio}}_{B(j)}+\beta^{\text{E24}}_{e_{24}(j)}+\beta^{\text{E48}}_{e_{48}(j)}+\beta^{\text{month}\times\text{Subregion}}_{m(j),\,R(j)}\right\}, \qquad (4.1)$$

in which j is the data record index, $\hat{\mu}_j$ is the fitted catch of coral trout (several species combined) for record j, d_j is the number of dories, x_j is the number of excess crew (crew in excess of the number of dories, zero if number of crew is less than or equal to number of dories), I is the indicator function (so that $I(d_j = 0)$ is equal to 1 if $d_j = 0$, and 0 otherwise), and the β's are parameters estimated by the model. Superscript "vessel" specifies the vessel efficiency parameters (one for each vessel), "Subbio" specifies Subbioregions (again one parameter for each Subbioregion), "E24" stands for categories of tropical cyclone energy lagged by up to 24 months, "E48" categories of cyclone energy lagged up to 48 months, and "month × Subregion" stands for combinations of month and Subregion (one parameter for each combination). Subscript $v(j)$ is the vessel that fished in record j, $B(j)$ is the Subbioregion in which it fished, $e_{24}(j)$ and $e_{48}(j)$ are the cyclone energy categories using a 24-month and 48-month lags respectively (see below), $m(j)$ is the month (combination of year and month, to produce a monthly time series), and $R(j)$ is the Subregion.

In the term involving $\beta^{(1)}$ in (4.1), the variate $d_j - 4$ is used instead of d_j for reasons of numerical stability; this change does not alter the structure of the model. The value of four dories is simply a typical number of dories used in the fishery. The term involving $\beta^{(2)}$ is an acknowledgement that a value of zero dories denotes fishing from the primary vessel and so should not correspond to zero fishing activity.

Lagged tropical cyclone energy was defined as a sum of monthly energy input to each six-minute grid square. For the 24-month lag, it was the sum of energies from cyclones that struck in the month the catch was taken and in the previous 23 months; any cyclone that occurred more than 24 months previously was not counted in the 24-month lag. Energy values were calculated for lags of 12, 24, 36 and 48 months. Initially, all of these lags were put into the analysis, but the 12-month and 36-month lags turned out to be only marginally statistically significant and were omitted (see Results section below).

Also it was not known what sort of functional form to expect for the dependence of catch rates on tropical cyclone energy inputs. Therefore the cyclone energy was defined as a categorical variable ("factor" in statistical software terminology). There were eight categories, which resulted in seven parameters for each value of the lag (see Table 19). The categories were zero (no cyclones to affect that location), 0–0.02 petajoules, 0.02–0.05 PJ, 0.05–0.1 PJ, 0.1–0.2 PJ, 0.2–0.5 PJ, 0.5–1 PJ and 1–5 PJ.

Cyclone energy was included on the assumption that it did not relate to abundance of coral trout but only to catchability (how easy it is catch the fish, not how many there are). Therefore, including it in the analysis would provide more accurate monthly time series of abundance, which were given by the $\beta^{\text{month}\times\text{Subregion}}_{m(j),\,R(j)}$ coefficients.

We remark that it would have simplified the subsequent analysis if the social learning effect of fishing on coral trout catch rates (see Chapter 2) could have been included in the catch-rate analysis. That was not possible because the social learning effect depended on the depletion ratio of the fish population, which was not known *a priori* and was only available from the population dynamic model itself. Therefore, social learning had to be included in the population dynamic model instead (see Chapter 6).

After all the parameter estimates had been found, they were summarised in order to facilitate presentation of the results. For example, monthly time series were converted to annual time series to show catch rates, and a single time series for the whole fishery was created by averaging over Subregions. These summaries were created by taking weighted averages of the series of parameter estimates.

The important aspect of the weighted averages was that a fixed weight was applied to each series; the weight did not change within a series. For example, to create an abundance estimate for all Subregions combined, each Subregion g was assigned a weighting factor w_g, which satisfied $\sum_g w_g = 1$, and then the abundance time series $Y_{g\,t}$ were combined to create the overall series

$$Y_t = \sum_g w_g \, Y_{g\,t} \,,$$

where t is the time index (either month or year). The weights w_g did not change with time. This approach avoids the problems described by Walters (2003) whereby if the weights change with time the overall time series is biased.

Walters' comments were particularly aimed at fisheries in which fishers travelled further offshore to new fishing grounds as the catch rates in existing fishing grounds declined. Then the overall catch rate would hold up even though the population was being depleted. In the context of coral trout, this problem could occur if fishers had heavily fished the accessible Subregions from Cooktown to Mackay, and then transferred their fishing effort to the Far Northern Subregions far from ports (Cape York, Lockhart River, and Princess Charlotte Bay) and further offshore to the Swains. Actually there is little sign that this has happened: the catch in the Far Northern Region is still small; although it sometimes increases after a tropical cyclone hits other Regions, it decreases again when the other Regions recover (see section 4.5). Even if the pattern of consistent expansion into the Far Northern Region did occur, the time-invariant weighting method used here would account for it and would avoid introducing a bias into the overall catch-rate time series.

We have not gone as far as Walters in that we have not weighted the Subregions by any measure of inferred population size or habitat area. We have preferred to calculate the weights based on fish that the fishery has actually found, and not infer any sizable population of fish that fishers have never seen.

The weight assigned to each series was proportional to the total fitted catch of the series, so that series providing higher precision about catch rates were assigned higher weights. In the example of amalgamating the Subregions, the weight assigned to a Subregion was proportional to the total catch taken in that Subregion over the history of the logbook database. This downweighted the Far Northern Region whose catch rates were, in any case, subject to substantial experimental error from low sample sizes.

Table 19: Parameters used in the generalised linear model for catch rates, with fitted values given by equation (4.1). The "Form" field is "A" for additive effect, and "M" for multiplicative effect. Due to lack of knowledge of the expected relationship between cyclone energy input and catch rate, cyclone energy was specified as a factor with eight levels.

Effect	**Form**	**Description**	**Parameters**
Dories	A	Number of dories	1
Zero dories	A	Adjustment for zero dories	1
Excess crew	A	Number of crew in excess of number of dories	1
Vessel	M	Vessel efficiency	529
Subbioregion	M	Effect of Subbioregion on abundance	45
Energy 24	M	Cyclone energy, linear decay over 24 months	7
Energy 48	M	Cyclone energy, linear decay over 48 months	7
Month	M	Monthly time series for each Subregion	2287
Total			**2878**

4.6.2 Results and discussion

Most of the important results are shown below in graphical form, as they are too numerous to show in tables. Table 20, however, shows the parameter estimates $\beta^{(1)}$, $\beta^{(2)}$ and $\beta^{(3)}$ pertaining to number of dories and number of crew. The value of $\beta^{(1)}$ of 0.1981 is slightly less than the value of 0.25 (the reciprocal of the reference number of 4 dories in equation (4.1)) that we would expect if fishing power were exactly proportional to the number of dories. This difference may result from the importance of the skill of the skipper and the dory operators, and possibly from some lack of completeness in accurately reporting the numbers of dories and crew employed in every fishing trip. Alternatively, if fishing takes place from the primary vessel, that could be considered an extra dory and the value 0.1981 would then be very close to the value 0.2 that we expect in this case, but our understanding is that fishing from large primary vessels is not common. The low value of $\beta^{(2)}$ compared to $\beta^{(1)}$ implies either that fishing from the primary vessel is not very efficient and that it is better to employ dories, or that fishing commonly takes place from small primary vessels whether or not they bring dories on a particular fishing trip. The small but nonzero value of $\beta^{(3)}$ may be a reflection of occasionally inaccurate reporting of the number of dories when the number of crew members exceeds the reported number of dories.

Table 21 presents an analysis of deviance table for the cyclone energy parameters. In a generalised linear model, analysis of deviance takes the place of the more well-known statistical technique of analysis of variance. The principle is the same, in that effects explain various amounts of the total random variation and what remains at the end is assigned to the "residual" deviance and can be used to calculate F-statistics. An F-statistic has an expected value of approximately 1, and values greater than 1 denote effects that explain more variation than we would expect by chance.

Nominally, the F-statistics for omitting the time lags of 12 months and 36 months are both highly statistically significant (values more than 5, with 7 degrees of freedom in the numerator and a very large number of degrees of freedom in the denominator), but given the large size of the data set these were not regarded as practically significant. Generally, the larger the data set the easier it is to prove that an effect is statistically significant, but statistical significance does not mean that an effect is physically meaningful. Also, the data records were not fully independent, because a single cyclone affected many different catch records. Therefore we regarded the 12-month and 36-month effects as non-significant, and omitted them.

The F-statistics for the 24-month and 48-month effects are large and establish a significant effect of tropical cyclones on coral trout catch rates. These effects are plotted in Figure 20. As expected, the coefficients for the 24-month lag decrease with increasing energy input, showing that tropical cyclones result in a fall in catch rates on this time scale.

Table 20: Parameter estimates and standard errors for number of dories and excess crew. Standard errors are very small due to the large size of the data set.

Parameter	Effect	Description	Value	S.e.
$\beta^{(1)}$	Dories	Number of dories	0.1981	0.0006
$\beta^{(2)}$	Zero dories	Adjustment for zero dories	0.0739	0.0037
$\beta^{(3)}$	Excess crew	No. of crew in excess of no. of dories	0.0468	0.0008

Table 21: Analysis of deviance table for different time lags (in months) of tropical cyclone energy inputs. The table shows, for each set of lags, the difference in deviance from the previous row, the difference in degrees of freedom, the residual deviance, the residual degrees of freedom, and the F-statistic.

Lags fitted	Diff. dev.	Diff. d.f.	Resid. dev.	Resid. d.f.	F
12, 24, 36, 48			5275224	265634	
24, 36, 48	809	7	5276033	265641	5.820
24, 48	730	7	5276763	265648	5.251
24	7993	7	5284756	265655	57.484
None	3451	7	5288207	265662	24.782

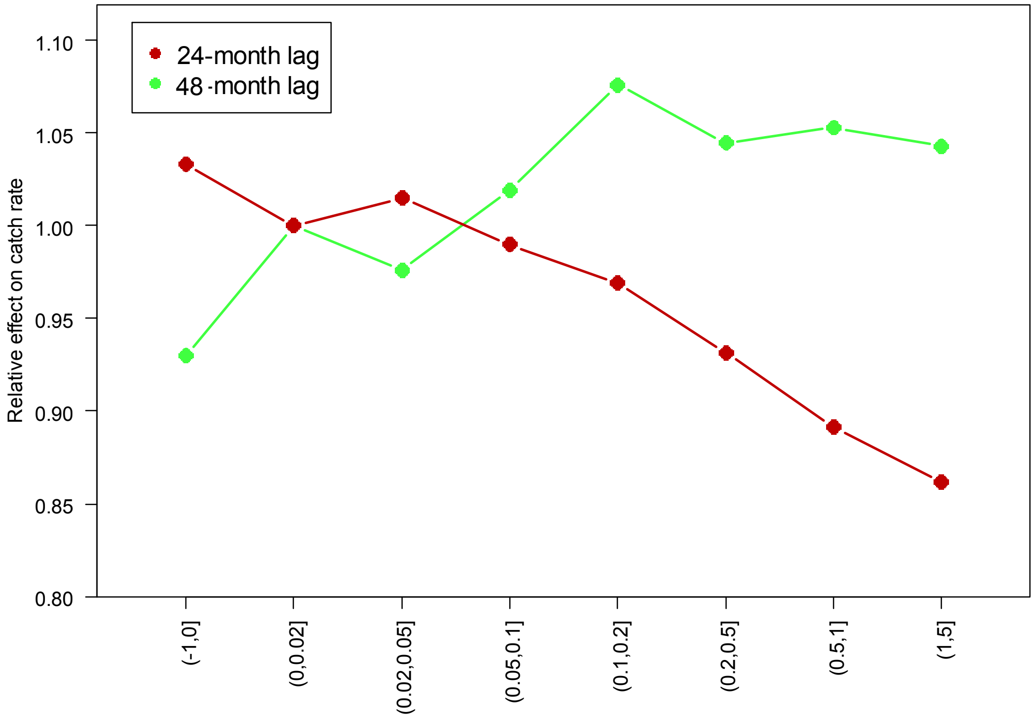


Figure 20: Coefficients for tropical cyclone energy inputs for time lags of 24 and 48 months. The x-axis shows categories of time-averaged cyclone energy inputs, in PJ.

A remarkable result is that the coefficients for the 48-month lag increase, indicating that catch rates bounce back strongly on that time scale. This finding offers support to the hypothesis put forward in section 3.1 that coral trout catch rates fall when a cyclone deprives the forage fish of hiding places, but bounce back when the coral recovers and predators (including coral trout) deplete the stocks of forage fish that had become easier for them to catch.

Judging from plots of catch size (Figure 18) and comments from fishers, the actual effect of tropical cyclones on the fishery is a good deal larger than that found here, which is a maximum fall of only about 10% (the product of the 24-month and 48-month coefficients at the highest energy level in Figure 20). This may be due to the limitations of the cyclone analysis in Chapter 3, which does not account for swells that can be generated by high winds hundreds of kilometres away. Future analyses of cyclones may predict bigger falls in catch rates, which would be more in line with the catch-size plots and fishers' experiences.

Catch rates by Subbioregion, averaged over all years, are plotted in Figure 21. Within each Subregion, the Subbioregions begin inshore (codes beginning with "RE") and proceed to mid-shelf and then to the outer reefs (codes beginning with "RA").

Inshore locations (RE Subbioregions) in general have lower catch rates, abundant bar-cheek coral trout (*Plectropomus maculatus*) (see section 4.1) and less commercial fishing than offshore (see Figure 19). In outer-mid-shelf locations (RB, RC and RG Subbioregions), common coral trout *P. leopardus* "comes into its own" as Ayling and Ayling (2000) state, and catch rates are relatively high. Outer-shelf catch rates (RA Subbioregions) are lower than the mid-shelf ones (RB and RC Subbioregions) in the Far Northern Region (Cape York, Lockhart River, and Princess Charlotte Bay Subregions), but are higher further south (Townsville and Mackay Subregions). In the Far Northern Region the RA Subbioregions contain the true "barrier" and ribbon reefs which are long and thin, aligned north–south with only small gaps between them, and are not the preferred habitat of *P. leopardus* (Ayling and Ayling, 2000). In the more southern Regions the reefs in RA Subbioregions are more detached.

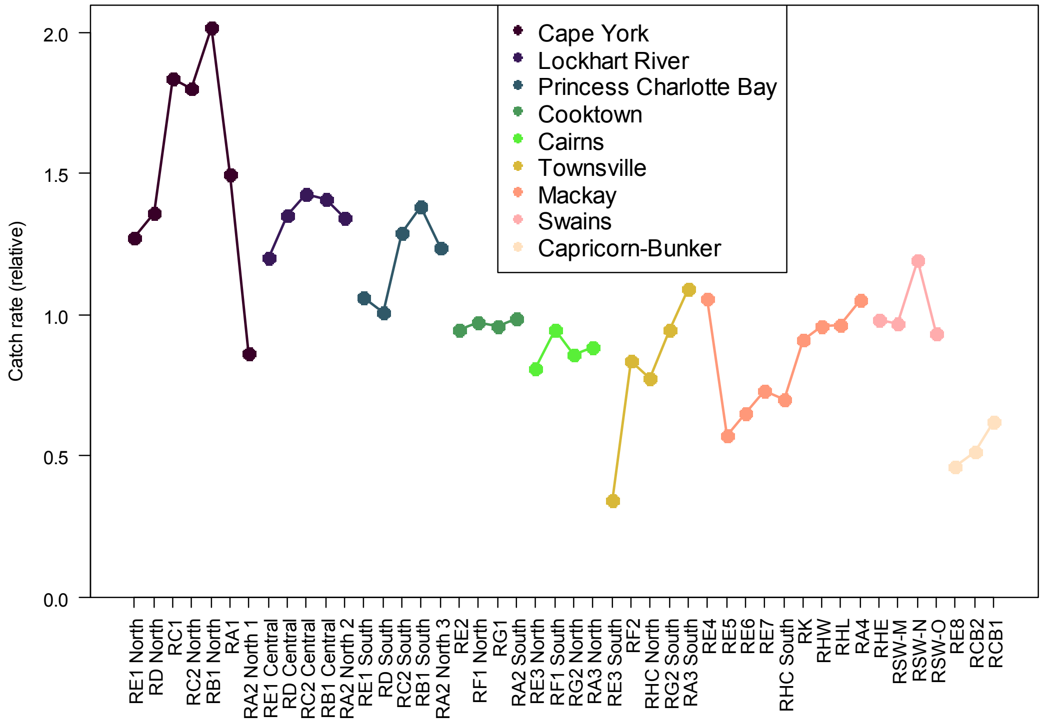


Figure 21: Standardised commercial catch rates by Subbioregion, averaged over all years, measured in non-dimensional units such that the overall average catch rate is equal to 1.

Still on Figure 21, catch rates show a decreasing trend from north to south from Cape York to Cairns. We ascribe this to the effect of fishing, and especially social learning by coral trout which quickly learn not to take bait when the population is fished (see Chapter 2). Fishing is still light in the Lockhart River and Princess Charlotte Bay Subregions (Figure 18), and habitat characteristics are much the same as evidenced by the Bioregions, most of which stretch over all three Subregions (see Figure 6 in Chapter 1, page 14). Therefore it is not reasonable to suppose that coral trout abundance is proportional to catch rate and that abundance in Lockhart River and Princess Charlotte Bay has fallen by the same proportion as catch rate has. The social learning explanation is supported by the findings in Chapter 2 and by fishers' observations and fishing strategies (described in Chapter 1).

Over the Cairns, Townsville and Mackay Subregions the catch rates don't change much from north to south. We assume that, under moderate levels of fishing, coral trout in these locations have learnt not to take bait. This assumption roughly matches the finding in Chapter 2 that the catch rate levels out after the population has been depleted by about 25%.

In the Swains, catch rates are higher than in the Townsville and Mackay Subregions. This may be due to lower fishing intensity because fishers have to travel further to reach the Swains, and possibly also to higher productivity associated with high tidal ranges and the presence of classical crescentic reefs.

The Capricorn–Bunker Region has the lowest catch rates of all the Regions. This is probably because the fishery in this Region is a mixed fishery that does not especially target coral trout.

The monthly series of catch rates, averaged over the whole fishery, is plotted in Figure 22. It shows that tropical cyclones affect the catch rate very soon after they strike, usually within one month. Catch rates for recent years show that life was especially tough for the commercial fishers in mid-2011, shortly after Cyclone Yasi, which itself came only two years after Cyclone Hamish. The standardised catch rates shown in Figure 22 take account of tropical cyclones to some extent, but the standardised catch rate in mid-2011 was still less than half what it was in 1995. The commercial fishery appeared to be recovering in 2013.

It is clear from Figure 22 that the tropical cyclone energy inputs have explained only a fraction of the impact of cyclones, because the effects of cyclones are still so obvious and so large in the standardised catch rates. Better cyclone-related input variables are required.

Any effect of the 2004 restructure on catch rates is unclear. Some stakeholders have suggested that catches may have been over-reported in the lead-up to the restructure date, in order to establish rights to quota allocations, and may have been under-reported after the restructure, in order to save on quota lease costs. The catch-rate data do not support this hypothesis, as catch rates from analysis of the logbook data are generally higher after the restructure than before, which is the opposite of what the hypothesis predicts. On the other hand, apparent catch rates increased shortly after the Investment Warning in May 1997; but this was also the recovery period from Cyclone Justin, and it remains difficult to explain the fall in apparent catch rates in the lead-up to the restructure. The only biological explanation we can offer for this fall is that it may be related to a multi-year ecosystem cycle involving balances between high-level predators (including coral trout) and forage fish.

The annual catch rate series over the whole fishery is plotted in Figure 23. It averages the information in Figure 22 over calendar years, and also shows the result of excluding tropical cyclone wind energy from the catch rate analysis.

The adjustment for tropical cyclones has performed very poorly. The only practically useful outcome is some smoothing out of the effect of Cyclone Justin from 1996 to 2000. The tropical cyclone adjustment achieved little smoothing of the effects of any cyclone other than Justin. Again it is clear that a better input than wind energy is required, possibly involving swell height.

The lack of accounting for cyclones, unfortunately, greatly limited the usefulness of the final catch rates to the population dynamic model, and necessitated downweighting of their contribution to the stock assessment (see Chapter 7).

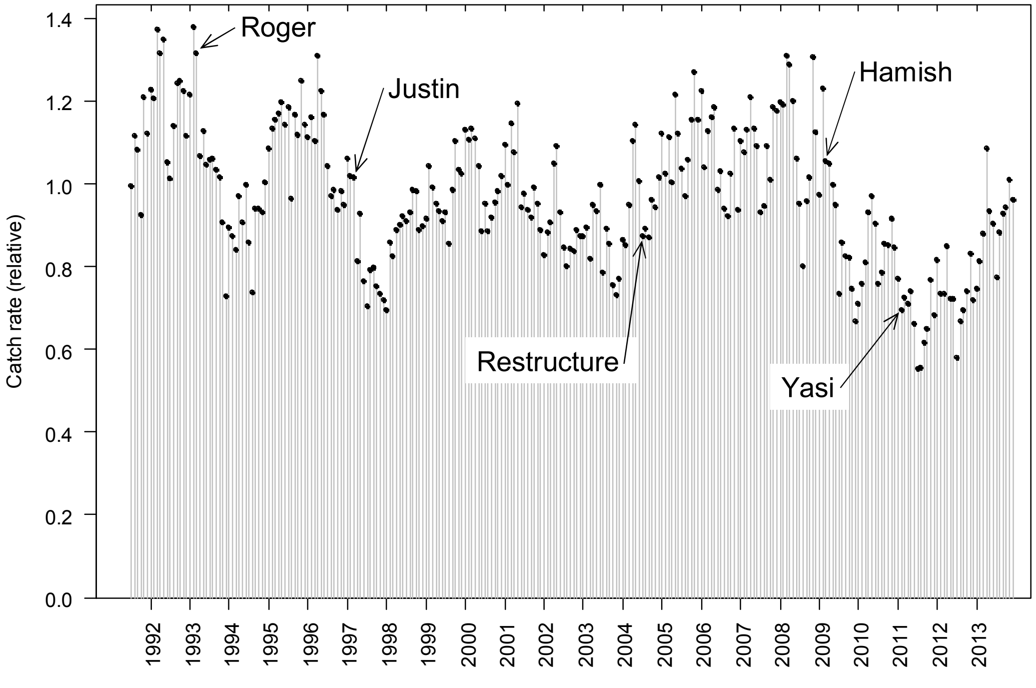


Figure 22: Monthly standardised commercial catch rates, averaged over all Subregions. Major events are labelled in the months in which they occurred.

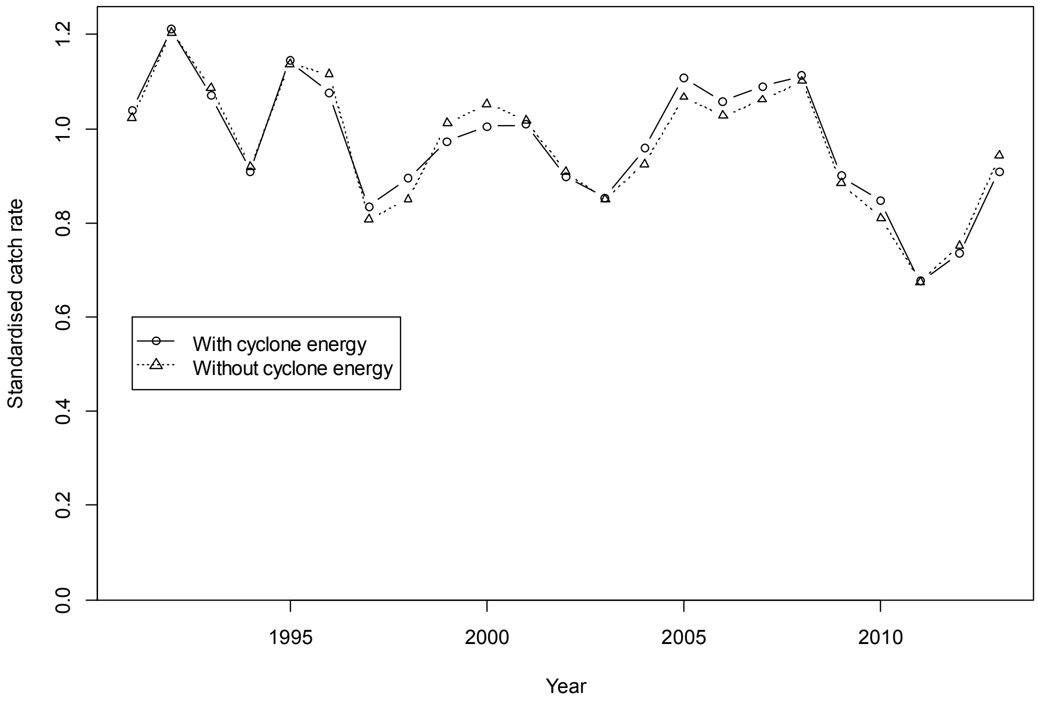
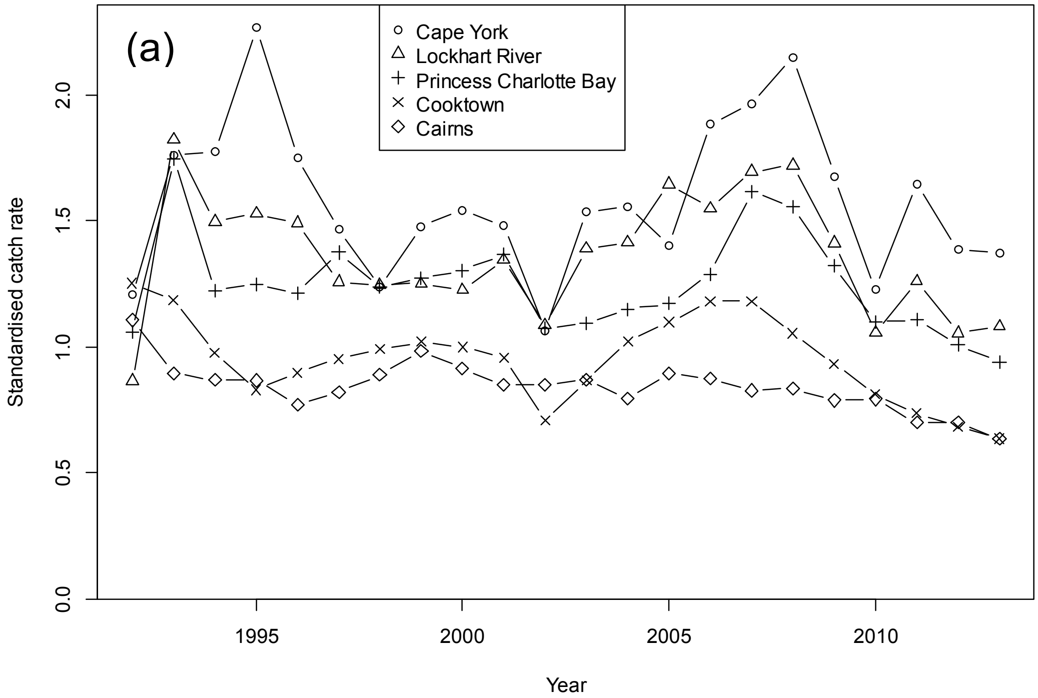


Figure 23: Annual series of standardised catch rate, by calendar year, averaged over all Subregions, also showing the results of analysis excluding tropical cyclone wind energy as an input variable.

Annual catch rates by Subregion are presented in Figure 24, and these were used as input to the population dynamic model in Chapter 6, although downweighted as mentioned above. The Townsville, Mackay and Swains Subregions all show a low point in 2011 and a recovery in 2012 and 2013. Over the few years leading up to 2011, all regions show declines. For most regions the 2011 catch rate was the lowest since the logbook database began. Many of the northern Subregions show a low point in 2002, and southern Subregions in 2003. The reasons for these are unknown. There was no major tropical cyclone at that time. The low catch rates may have some connection to the coming 2004 restructure of the fishery, although it is not clear how. Alternatively, as mentioned above for the fishery-wide catch rate, they may be related to complex ecosystem dynamics.

The vessel efficiency factors that came out of the catch-rate analysis are plotted as a histogram in Figure 25. The histogram shows a mode at a value slightly less than 1, which represents an average vessel that specialises in catching coral trout. Values greater than 1 represent exceptionally efficient vessels. There is a wide distribution of vessels with efficiency factors between zero and the mode. Many of these may be vessels that do not specialise in coral trout.

It was not necessary to omit vessels with low efficiency factors from the analysis, because the quasi-Poisson model automatically downweighted them.



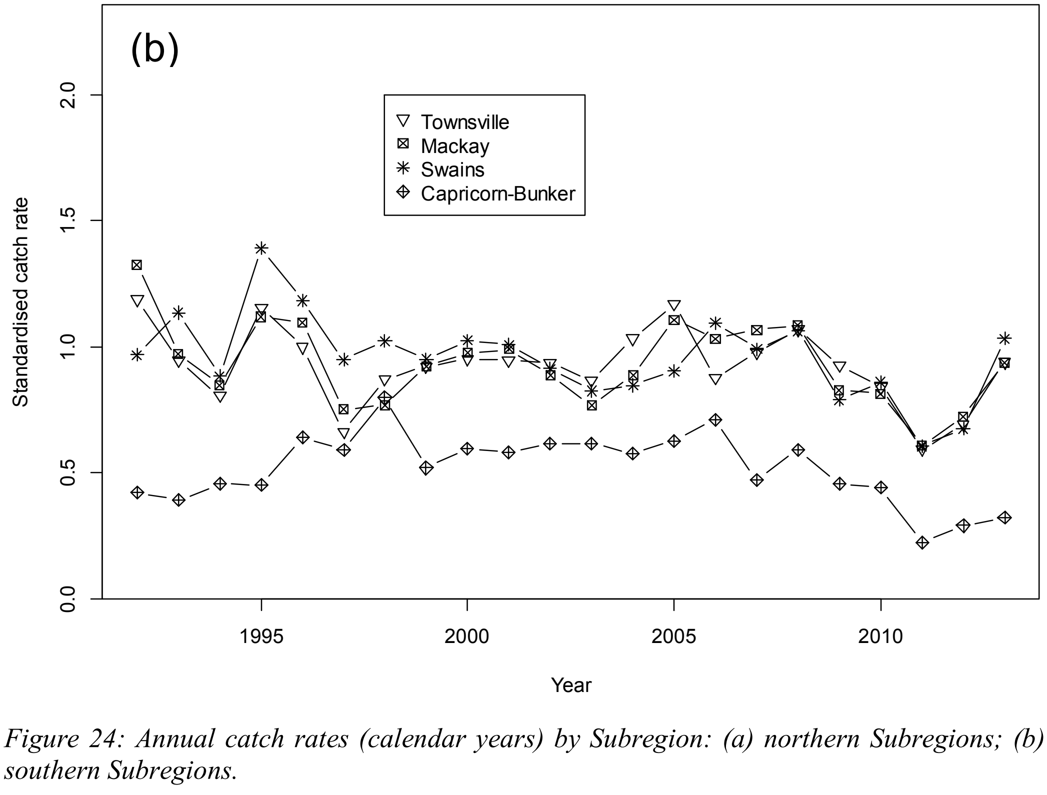


Figure 24: Annual catch rates (calendar years) by Subregion: (a) northern Subregions; (b) southern Subregions.

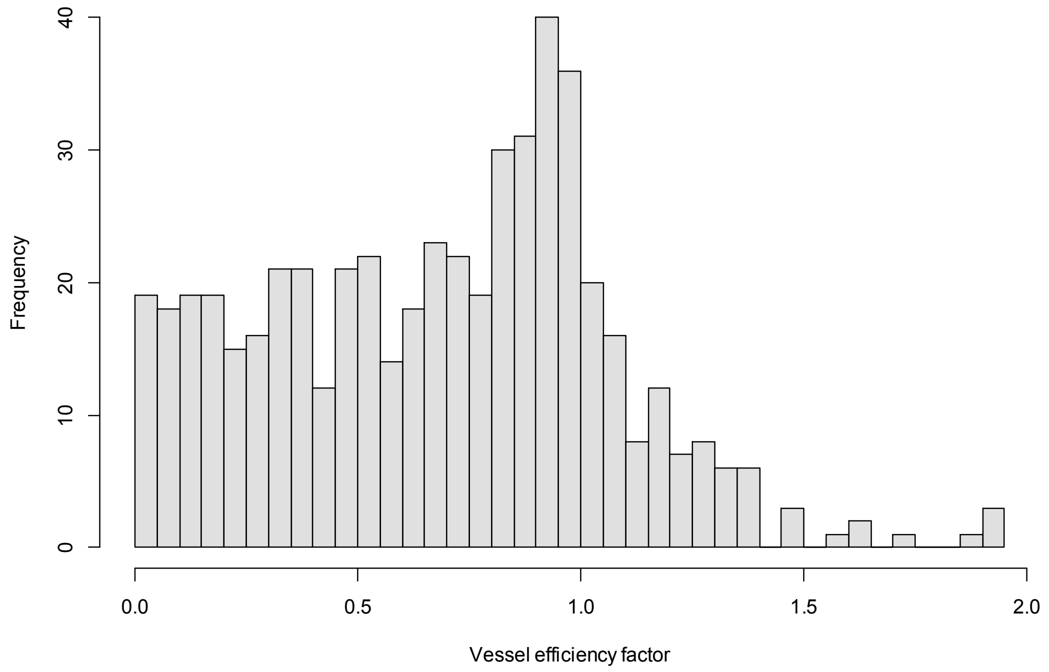


Figure 25: Histogram of vessel efficiency factors from the commercial catch-rate analysis. The quasi-Poisson model automatically downweights the less efficient boats which may not specialise in coral trout; hence they do not need to be omitted from the analysis.

4.7 AIMS underwater visual survey data

Underwater visual survey data from the Long Term Monitoring Program run by the Australian Institute of Marine Science (AIMS LTMP) provided abundance data for coral trout. Although they provided absolute abundance measures in number of fish per hectare, they were used only as relative abundance measures, due to differences in protocol from the GBRMPA underwater visual surveys. One major difference is that AIMS LTMP records stripped out the "young of the year", i.e., the newly born fish. Also the surveys were conducted on the north-east corner of each reef, whereas the GBRMPA surveys were conducted mainly on the back reef slope (north-west side of the reef). The AIMS LTMP sampling protocol is described in detail by Halford and Thompson (1994).

The surveys were carried out on groups of reefs called "Sectors". These were converted into the Subbioregions used in the stock assessment. Annual trends in abundance over time for each Subbioregion are plotted in Figure 26.

The trends show very large amounts of variation, with abundance at most locations changing by a factor of five or more over the course of the time series. The early years especially appear to be anomalously high in many locations. For use in the stock assessment, we omitted all data before 1995.

Standard errors of the underwater visual survey data are also high, which is typical for this kind of data.

4.8 Information transferred into the stock assessment

The following information was carried forward into the stock assessment from this chapter:

- Justification for omitting inner-shelf Bioregions from the assessment, as the assessment is for common coral trout Plectropomus leopardus and concentrates on the commercial fishery (section 4.1)

- Reconstructed harvest size history (section 4.5)
- Time series of standardised catch rates for each Subregion, as indices of abundance, suitably downweighted to compensate for the inability of current methodology to deal with changes in catch rates caused by tropical cyclones (section 4.6)

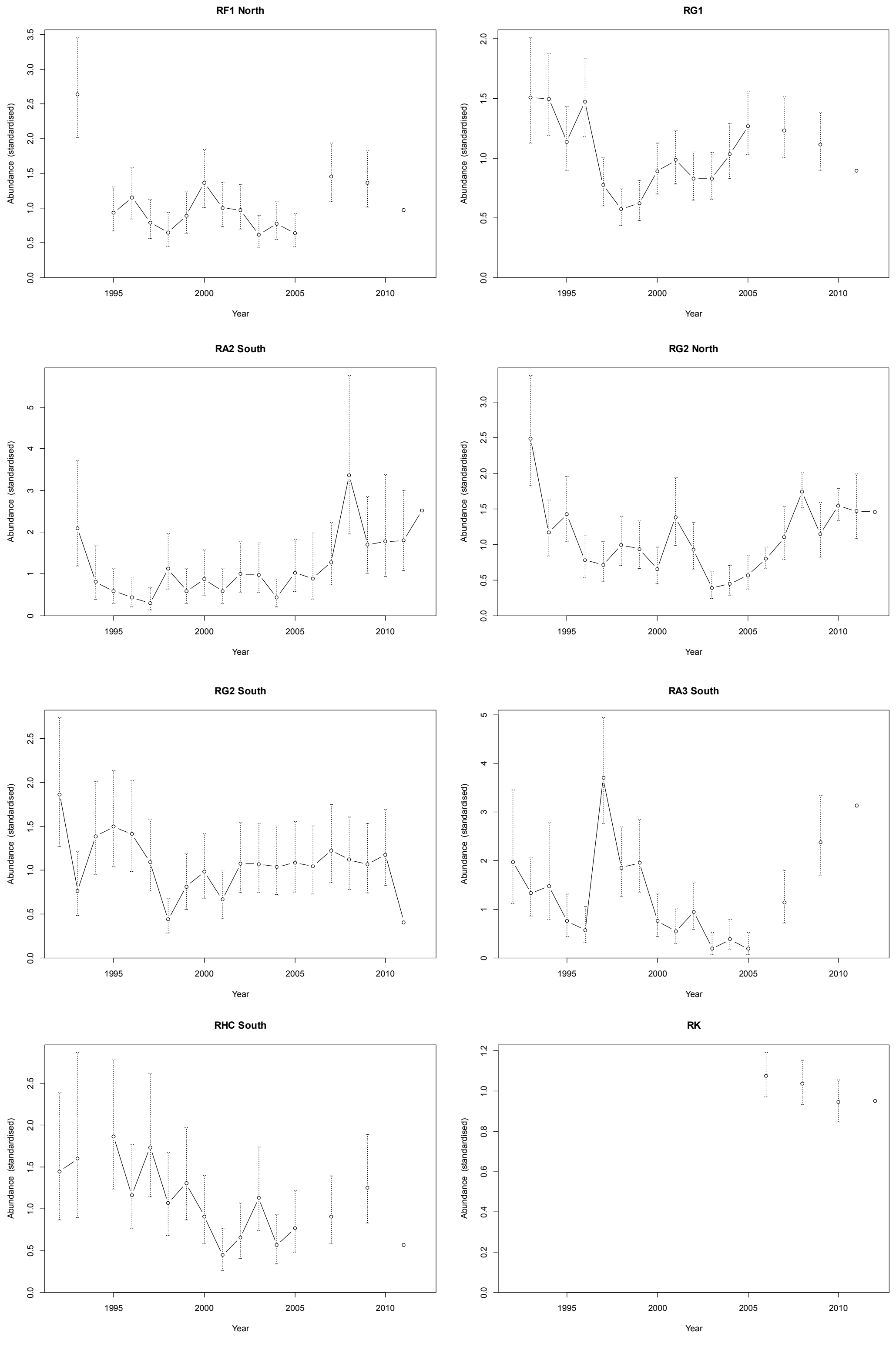


Figure 26: Annual time series of abundance estimates of P. leopardus from AIMS underwater visual surveys, for each Subbioregion, scaled to average to 1. The error bars represent one standard error. The scaling has the side effect that the final point in each series is a reference point and so is considered exact (standard error of zero); the standard errors of the other points are standard errors of the difference between them and the reference point.

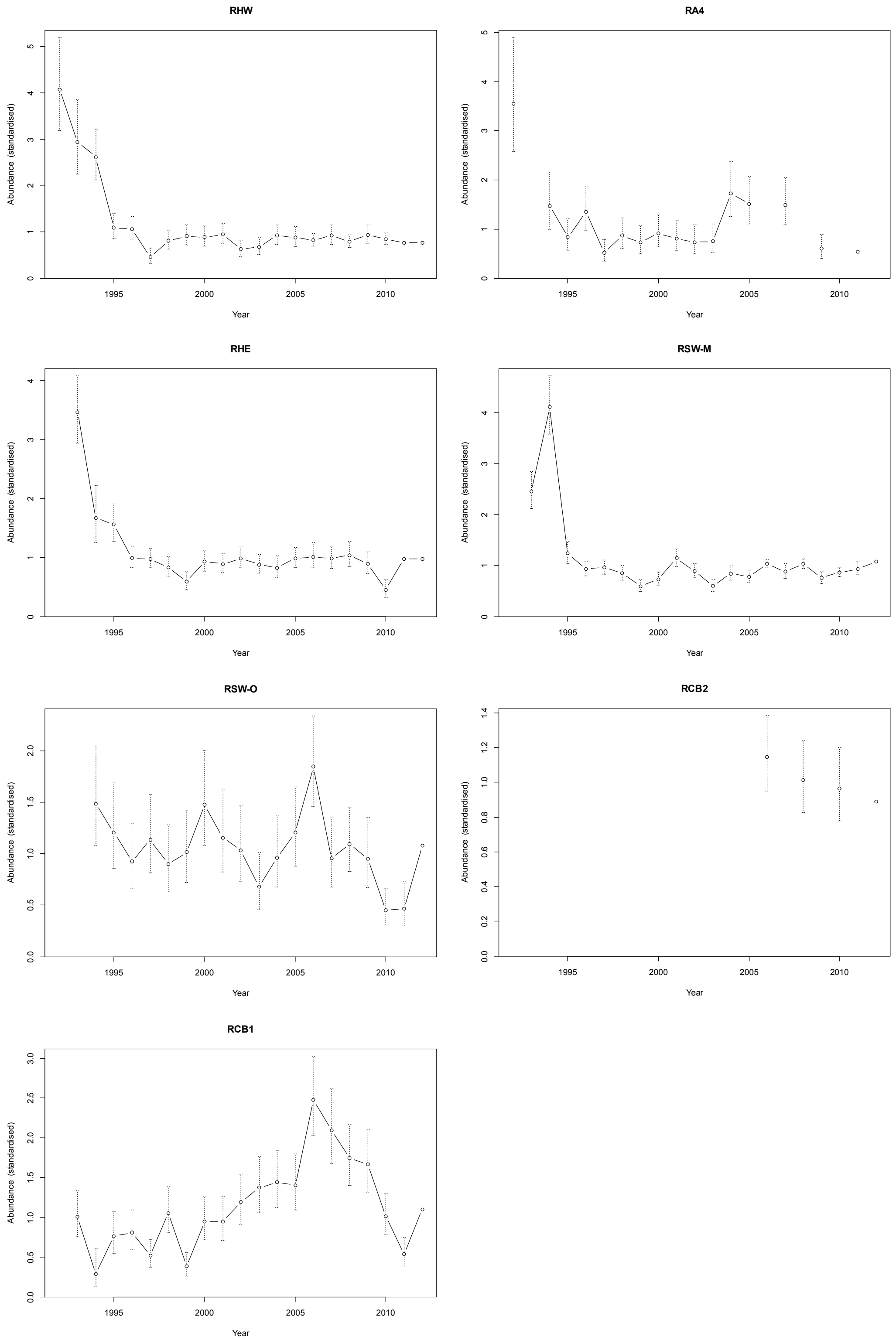


Figure 26, continued from previous page.

5. Population estimate from underwater visual surveys

5.1 Background

A large series of underwater visual surveys on more than 200 reefs was undertaken from 1983 to 1986, funded by the Great Barrier Reef Marine Park Authority (GBRMPA) (Ayling and Ayling, 1986). They were all conducted by Dr Tony Ayling and Dr Avril Ayling of Sea Research, and the capability of the methodology to count all of the fish present in a transect was analysed in detail by Ayling et al. (2000).

A survey of this size provided a unique opportunity to calculate a baseline population size of common coral trout on the Great Barrier Reef. This is extremely useful to anchor the results of the stock assessment, and can help to compensate for the very poor correlation between catch rates and abundance caused especially by tropical cyclones (sections 4.5 and 4.6) and social learning (Chapter 2).

The 1983–86 surveys were carried out in lots of ten transects, each 50 m long by 20 m wide. The divers swam out for 50 m in a straight line along a randomly laid tape, counting the fish within 10 m on one side, and then swam back and counted the fish within 10 m on the other side. Later surveys standardised the methodology to 50 m × 5 m transects, in which the divers swam in only one direction and counted all fish within 2.5 m either side of the tape. These surveys were believed to count all of the sizable fish on the transect, whereas the 50 m × 20 m transects missed some because the divers didn't see them; Ayling et al. (2000) calculated a ratio of 0.358 to convert 50 m × 20 m counts to 50 m × 5 m ones, not the value of 0.25 that one would expect if the larger transect counted all the fish.

The surveys were carried out mainly on the back reef slope, which had the highest abundance of coral trout (slightly more than the front). Some were carried out on the front reef slope, which allowed calibration of the abundance between the front and the back. The lagoon of one reef was surveyed and the abundance there was found to be much less than on the back reef slope. Submerged parts of reefs were not surveyed because the divers could only descend to 10 m depth without resorting to decompression dives which would have been very time-consuming and potentially dangerous.

5.2 Methods

Mapping data for reefs was provided in electronic form by GBRMPA. This allowed calculation of the amounts of front-slope, back-slope, patch-reef and submerged habitat in each Bioregion (see Chapter 1 for a description of Bioregions). The mapping data contained a "dry reef" database which showed the dry or "hard" reef that emerged above the water at high tide. It also contained a "wet reef" database which showed the potential habitat below the water.

Reef slope habitat was considered to be the area from the edge of the dry reef to a depth of 30 m, which was about the greatest depth found in the reef-slope surveys. The survey data contained estimates of the depth at each end of the transect, so we were able to extrapolate each reef linearly to find the width of habitat at which the depth would have been 30 m. We averaged the reef slope width over all reefs surveyed in each Bioregion. When no reefs had been surveyed in a particular Bioregion, we used a value of 50 m as a typical reef slope width (Tony Ayling, personal communication).

Patches of dry reef were also assumed to give rise to reef slope habitat, on the basis of their obvious popularity with fishers; the most intensively fished grid squares in the commercial logbook data were those with large amounts of reef patches. The width of reef-slope habitat on reef patches was assumed to be the same as on the back slope of the reef.

Submerged reef habitat was considered to be the area of the wet-reef shapes, minus the reef slope and the dry reef. Calculation of all these areas was programmed in the software *R* (R Core Team, 2013) from the electronic mapping data. The density of coral trout in the submerged habitat was subject to some uncertainty. It was assumed to be a fixed proportion of the density on the back reef slope. This proportion came from the density in the lagoon for the one reef (Sanctuary Reef, 22-109 in the Swains, Bioregion RSW-M) whose lagoon had been surveyed. This was considered to be a conservative figure, as submerged reef is thought to be more productive than lagoons, but it was also found to be reasonable by examination of commercial logbook data for grid squares that contained substantial amounts of submerged reef.

The density of coral trout (fish per hectare) in the survey data was scaled up by the habitat area in each Bioregion, to arrive at population estimates by Subbioregion. Density was assumed to be constant within each Bioregion. The length of every fish sighted during the underwater visual surveys was recorded, allowing population density estimates to be restricted to legal-sized fish (38 cm total length or above).

Estimates of fish density can be subject to biases. For underwater visual surveys of coral trout, potential sources of bias have been investigated thoroughly and correction factors have been calculated where necessary (Ayling et al., 2000; Mapstone and Ayling, 1998). For coastal rocky reefs in Tasmania, Edgar et al. (2004) compared underwater visual surveys to capture-resight methods which are themselves subject to substantial biases, especially when the spatial extent of the range of the marked fish is not well defined. Nevertheless, their results indicate that biases for easily detectable predators such as legal-sized coral trout are low.

5.3 Results

Ayling and Ayling (2000) state that the abundance of coral trout is slightly less on the front reef slope than on the back. They based this statement on transects laid down the reef slope. When the abundance was measured with respect to horizontal area instead of area down the slope, there was no significant difference between the front and the back slope, because the front slope was usually steeper than the back. The difference between front and back reef slope appears to be not so much a difference in density of coral trout per horizontal hectare, as a difference in width of the habitat, with the front being steeper and hence narrower than the back, bearing in mind the cut-off of 30 m depth for reef slope habitat.

The estimate of the ratio of density of fish on submerged reef to density on reef slope, from the one reef that had been surveyed, was 0.1595167. This value was also consistent with examination of logbook data records from grid squares with different amounts of submerged reef: a much higher value would have meant that fishers should have fished the submerged habitat more intensively, while a much lower value would have meant that fishers should have avoided fishing submerged habitat.

Some Bioregions were not included in the 1980s underwater visual surveys. Most of these were not important to the commercial coral trout fishery. The exception was the RHL Bioregion ("Hard Line" reefs) which was highly desirable to add into the results. The abundance of common coral trout *P. leopardus* on these reefs was assumed to be the average of the abundances from the two neighbouring Bioregions, RHW (reefs that were more sheltered) and RA4 (exposed outer reefs).

Habitat estimates are listed in Table 22, and population estimates of *P. leopardus* in Table 23. The Mackay Region contributes the most habitat and population, and also the highest proportion of the catch (see Figure 18 in Chapter 4, page 62). The population estimates are restricted to legal-sized fish (38 cm total length or above).

The analysis establishes a population size of about 5.34 million adult fish in the mid-1980s. Using the average weight of a legal-sized fish of 1.576 kg from section 1.5.3, this equates to an exploitable biomass of about 8,400 tonnes. Current catch levels are not much greater than they were in the mid-1980s, although they were much greater in the early 2000s. Current exploitable biomass could be expected to be less than the figure of 8,400 t, but not by a great amount.

The population estimates above are for habitat whose existence has been well established by fishing. There may be sizable populations of coral trout in other places, e.g. deep water, but underwater video surveys conducted by the Australian Institute of Marine Science (AIMS) indicate that there are not vast amounts of coral trout habitat in deep water (Michael Cappo, AIMS, personal communication).

To further make the stock assessment relevant to the commercial fishery, a set of high-catch Subbioregions (marked * in Table 23) was defined. The population dynamic model was restricted to these Subbioregions. The estimated 1980s exploitable biomass in the high-catch Subbioregions was about 7,700 t.

5.4 Information transferred into the stock assessment

The population dynamic model in Chapter 6 was designed to use the above results. The population densities from underwater visual surveys were used as *absolute* measures of abundance that specified the number of fish present per hectare of habitat. This differs from most stock assessment models which use abundance estimates only as *relative* estimates to compare abundance in one year to that in another year.

The habitat areas were also used. Recruitment was parameterised as a number of recruits per hectare, and was scaled up by the habitat area to produce the total number of recruits.

Green zones contributed to recruitment in blue zones by summing the egg production across blue zones and green zones within the same Subbioregion. Recruitment was assumed to have the same density (recruits per hectare) in green zones as in blue zones.

Table 22: Area of different types of coral trout habitat (in hectares), by Subbioregion and zoning (blue open to fishing versus green closed to fishing). The types of habitat are reef slope, reef patches, and submerged reef. Zoning is the current zoning, from July 2004.

Subregion	**Subbioregion**	**Blue area**			**Green area**		
		Slope	Patch	Subm.	Slope	Patch	Subm.
Cape York	RE1 North	2113	807	1052	218	124	283
	RD North	610	180	1223	3101	772	1284
	RC1	1107	0	12272	60	0	7115
	RC2 North	283	0	48599	1546	27	85487
	RB1 North	95	20	18836	727	47	17876
	RA1	1749	83	6651	996	828	4753
	RA2 North 1	0	0	0	1548	43	6306
Lockhart River	RE1 Central	1291	714	550	847	298	1857
	RD Central	3466	4520	42103	813	470	18303
	RC2 Central	1827	1037	79715	85	0	25267
	RB1 Central	842	2	116582	148	63	12080
	RA2 North 2	2819	242	4704	1925	246	25645
Pr. Char. Bay	RE1 South	1605	804	414	403	151	133
	RD South	1116	895	697	266	27	3180
	RC2 South	5055	2704	35654	2360	3073	16406
	RB1 South	211	0	13430	946	161	1817
	RA2 North 3	1377	843	7862	931	497	7616
Cooktown	RE2	521	26	3844	649	646	398
	RF1 North	1448	943	1660	1253	442	775
	RG1	7013	5010	79341	1561	2008	24759
	RA2 South	3068	1603	25252	2294	1687	20922
Cairns	RE3 North	691	529	294	195	129	32
	RF1 South	102	35	120	77	36	145
	RG2 North	4782	3183	74944	1617	323	24729
	RA3 North	527	88	12133	392	46	11103
Townsville	RE3 South	597	346	597	80	17	16
	RF2	12	6	1169	0	0	261
	RHC North	810	1311	0	0	0	0
	RG2 South	5452	7756	116866	1874	1080	33999
	RA3 South	302	30	8516	426	13	9366
Mackay	RE4	2101	452	3080	293	83	414
	RE5	747	189	5637	59	40	2784
	RE6	925	309	2640	366	43	3641
	RE7	181	238	49	84	21	26
	RHC South	2045	885	911	510	253	54
	RK	6500	6151	39030	1640	1464	11133
	RHW	11146	9000	130798	5162	2989	38514
	RHL	14586	13114	46509	3869	3678	14528
	RA4	1886	1158	18557	604	11	10472
Swains	RHE	3153	581	22246	1198	840	5952
	RSW-M	5329	1094	52924	2184	506	20234
	RSW-N	657	31	8064	1012	164	6119
	RSW-O	1596	726	5688	1275	112	4023
Cap.–Bunker	RE8	825	333	341	90	43	513
	RCB2	82	0	5841	100	0	4042
	RCB1	1140	180	23659	1106	277	12544
Total		**103791**	**68159**	**1081055**	**46890**	**23780**	**496906**

Table 23: Habitat area (equivalent reef slope, ha), abundance (fish ≥ 38 cm total length per hectare) and population size estimates of common coral trout P. leopardus*, by Subbioregion and zoning. Cells labelled NS correspond to Bioregions that were not surveyed; bioregion RHL was important but not surveyed, and was assigned the average abundance of the two neighbouring Bioregions RHW and RA4. Subbioregions designated high-catch are marked *.*

Subregion	**Subbioregion**	**Habitat area**		**Abundance**	**Population size**	
		Blue	Green		Blue	Green
Cape York	RE1 North	3088	387	1.03	3169	397
	RD North	985	4078	5.95	5862	24275
	RC1*	3065	1195	9.38	28733	11200
	RC2 North*	8036	15210	10.38	83409	157875
	RB1 North	3120	3625	16.98	52967	61553
	RA1	2893	2582	5.92	17139	15293
	RA2 North 1	0	2597	8.71	0	22617
Lockhart River	RE1 Central	2093	1442	1.03	2148	1480
	RD Central*	14702	4203	5.95	87515	25017
	RC2 Central*	15580	4115	10.38	161715	42717
	RB1 Central*	19440	2138	16.98	330055	36304
	RA2 North 2	3811	6262	8.71	33188	54523
Pr. Char. Bay	RE1 South	2475	576	1.03	2540	591
	RD South	2123	800	5.95	12635	4764
	RC2 South*	13446	8050	10.38	139572	83558
	RB1 South	2353	1397	16.98	39950	23727
	RA2 North 3	3474	2644	8.71	30252	23020
Cooktown	RE2	1161	1358	8.49	9852	11529
	RF1 North*	2656	1819	9.23	24520	16792
	RG1*	24680	7518	13.52	333702	101657
	RA2 South*	8699	7318	8.71	75751	63724
Cairns	RE3 North	1267	329	0.00	0	0
	RF1 South	156	137	9.23	1443	1262
	RG2 North*	19919	5884	16.44	327526	96756
	RA3 North*	2550	2209	13.65	34823	30167
Townsville	RE3 South	1038	100	0.00	0	0
	RF2	205	42	NS	NS	NS
	RHC North	2121	0	6.04	12803	0
	RG2 South*	31850	8377	16.44	523697	137744
	RA3 South*	1691	1932	13.65	23087	26388
Mackay	RE4	3044	442	3.10	9442	1371
	RE5	1836	543	NS	NS	NS
	RE6	1656	990	NS	NS	NS
	RE7	427	110	NS	NS	NS
	RHC South	3074	772	6.04	18557	4659
	RK*	18877	4880	30.66	578719	149619
	RHW*	41011	14294	18.04	739948	257900
	RHL*	35119	9864	16.93	594434	166971
	RA4*	6004	2286	15.81	94922	36136
Swains	RHE*	7282	2987	16.36	119146	48879
	RSW-M*	14865	5918	35.55	528459	210390
	RSW-N	1974	2152	11.35	22408	24423
	RSW-O*	3229	2029	21.05	67984	42715
Cap.–Bunker	RE8	1213	215	NS	NS	NS
	RCB2	1014	745	NS	NS	NS
	RCB1	5094	3384	33.30	169647	112684
Total		**344396**	**149935**	**15.12**	**5341720**	**2130676**

6. Population dynamic model

6.1 Overview

The population dynamic model used in this assessment was a regional, age-structured, forward-prediction model. It was written in the software AD Model Builder (ADMB) (Fournier et al., 2012) and built on the general-purpose stock assessment model Cabezon (specifically, the Original Cabezon or OC model) (Cope et al., 2003). The source code for Cabezon was kindly provided by Dr André Punt of CSIRO and the University of Washington. Cabezon is also the name of a fish on the west coast of North America, for which this model was used, although later assessments of this species employed different software.

Cabezon calculates the number of fish of each age and sex in each year, and applies harvest rates (calculated from the recorded catch sizes) and the natural mortality rate to progress forward from one year to the next. It includes calculations of length-at-age and weight-at-age. A particular strength of Cabezon is the capability to include multiple "fleets" which can be either fishing fleets or scientific research surveys, all of which may have different age- or length-dependent vulnerability functions. Fishing is assumed to take place as a short pulse in the middle of each year. This does not exactly match the coral trout fishery, in which fishing takes place all year round, but because the coral trout are relatively long-lived we did not believe that the errors would be significant.

Cabezon model projections can be matched against abundance indices, age-frequency data and length-frequency data.

The software ADMB first estimates the model parameters by maximum likelihood, which is a long-standing and widely used statistical technique. It can then run simulations using Markov chain Monte Carlo (MCMC) to provide a random sample of potential parameter values. Confidence limits for the parameters can be constructed from the simulations.

Building on the Cabezon population model, the following additional capabilities were incorporated for coral trout:

- Regional structure: This took into account the qualitatively different Regions, Subregions, Bioregions and Subbioregions of the GBR, and the green zones (zones closed to fishing).
- Green-zone fishing parameter: This parameter was the ratio of the fishing intensity in a green zone to that in blue zones in the same Subregion. It was impossible to estimate from the available data, and, based on advice from industry and government, was set to 0.2.
- Absolute abundance measures from underwater visual surveys (UVS): Generally, abundance measures in stock assessment are only *relative* abundance indices which compare one year against another and don't provide information on the actual numbers of fish present. An *absolute* abundance measure specifies the actual density of fish in a population, in this case as a number of fish per hectare.
- Habitat area: The area of habitat (in hectares) of each regional population of fish provided a way to scale up the fish density (number of fish per hectare) into an estimate of population size (an absolute number of fish in a Population). The methodology for this is described in Chapter 5.
- Changes in zoning: The appropriate numbers of fish were transferred between green and blue zones in years when the zoning changed, according to the area of the rezoned habitat.
- Size limits: A reduced fishing mortality rate (the post-release mortality rate) was assumed to operate on fish that were below the minimum legal size (MLS). The model assumed that fishers released all undersized fish, but not all of them survived.
- Social learning (hook shyness): A social-learning parameter was introduced to quantify a coral trout population's ability to learn not to take bait when the population

is fished. Population dynamics were still determined by the actual fishing mortality rate, but the model's predicted fishery catch rates were those of a parallel population with the current year's fishing intensity scaled up by the social-learning parameter.

The above concepts are expanded upon in the following sections.

6.2 Regional structure

The regional structure for the model was based on the Reef Bioregions defined by the GBRMPA expert committees (see section 1.2 in Chapter 1, and Figure 6 on page 14). The Bioregions grouped together reefs with common habitat features. The model assumed that the virgin population density (number of coral trout per hectare) was constant within each Bioregion.

The Bioregions naturally divided the GBR into six Regions from north to south: the Far Northern Region, the Cooktown Region, the Cairns–Townsville Region, the Mackay Region, the Swains Region and the Capricorn–Bunker Region (see Figure 27). Every Bioregion except three was contained within a single Region: RA2 (Outer Barrier Reefs) covered both the Far Northern and the Cooktown Regions; RF1 (Northern Open Lagoon Reefs) covered the Cooktown Region and the Cairns Subregion; and RHC (High Continental Island Reefs) covered the Townsville Subregion and the Mackay Region.

Inspection of the commercial fishery logbook data showed that the intensity of fishing increased markedly from north to south. The far north of the GBR was relatively lightly fished, possibly because of the distance from port; Cairns was the nearest port from which live fish could be exported. Fishing intensity steadily increased as the location moved south to Townsville, and thereafter was roughly constant from Townsville south to the Swains.

Therefore two of the northern Regions were divided into Subregions within which the fishing intensity could be considered constant. The Far Northern Region was divided into three Subregions: Cape York (to 11.7 °S), Lockhart River (11.7 °S to 13.0 °S), and Princess Charlotte Bay (from 13.0 °S); and the Cairns–Townsville Region was divided into two Subregions, Cairns and Townsville (split at 18.1 °S). This also necessitated splitting many of the Bioregions into Subbioregions along the Subregion boundaries.

Finally, each Subbioregion was divided into two Populations, one containing fish in blue-zoned reefs (open to fishing), and the other containing fish in green-zoned reefs (closed to fishing).

The different levels of the regional structure are illustrated in Figure 28.

Reefs zoned yellow, where fishing was restricted to one dory per primary commercial vessel and one hook per dory, were counted as blue. Commercial fishers use only one hook per dory in any case, so this restriction did not affect them in practice. The restriction of one dory per primary vessel was a problem when a yellow reef was surrounded by green reefs and thereby isolated from blue reefs; then it was not feasible for a primary vessel to drop one dory at the yellow reef and the others at blue reefs. The only reef where this was known to occur was Old Reef (number 19-048), and the commercial catch returned from that reef was indeed lower than from other blue and yellow reefs in the vicinity.

When reefs within a Subbioregion were rezoned, the model transferred fish between blue and green populations according to the area of rezoned habitat. This allowed the model to cope with changes in catch rates caused by rezoning: if blue reefs were rezoned to green, fishers would have to operate within a smaller area which would be fished more intensively; hence catch rates would fall if either the total fishing effort or the total catch remained the same. The most notable example of this was the 2004 rezoning when the area of green zones increased from about 5% of the GBR to about 33%.

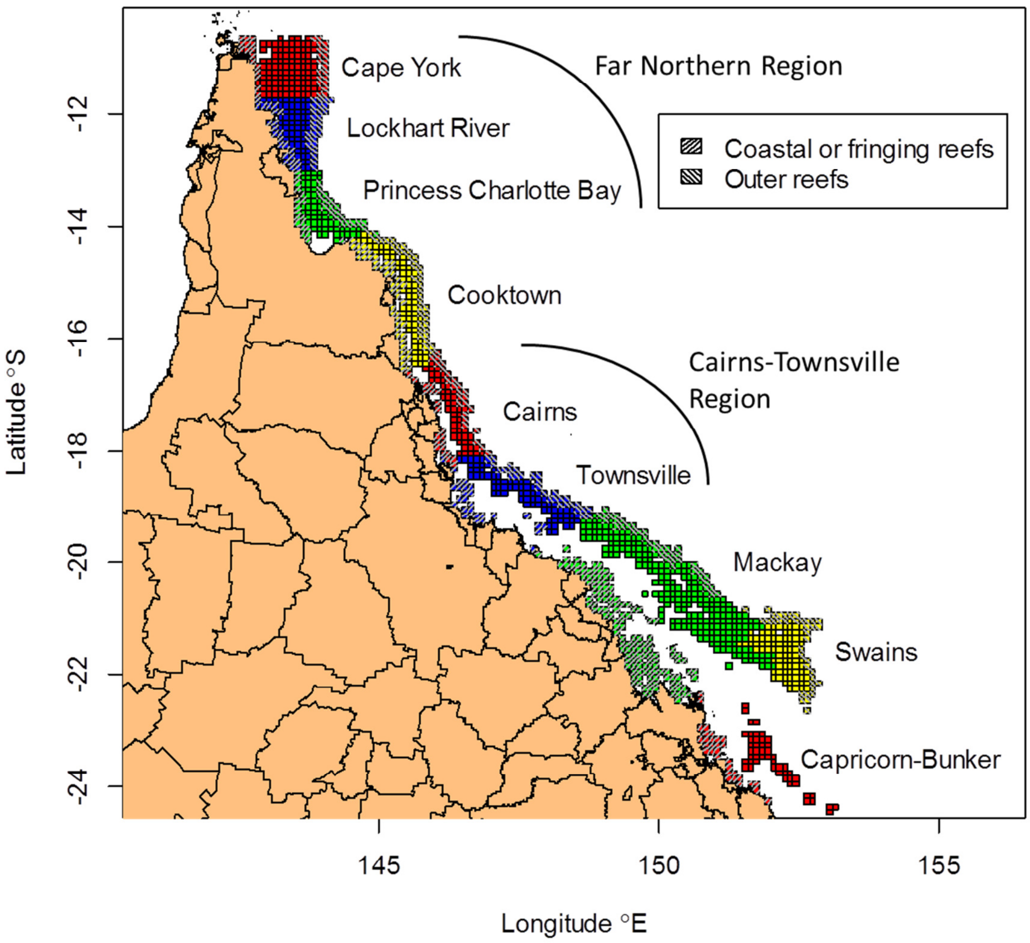


Figure 27: Reproduction of Figure 7: Regions and Subregions used in the stock assessment. Because the fishing intensity increases from north to south in the northern regions, the Far Northern Region is divided into three Subregions, and the Cairns–Townsville Region into two Subregions. The small squares are six-nautical-mile fishery logbook grid squares. Colours are chosen only to distinguish the Regions and Subregions, and have no other meaning. The Capricorn–Bunker Region was excluded from the stock assessment because fishers there did not strongly target coral trout, and underwater visual surveys showed dramatically different trends in coral trout abundance from the rest of the GBR.

This stock assessment includes only Subbioregions where substantial commercial catches of coral trout were taken. It omits Subbioregions in which targeted commercial fishing for *Plectropomus leopardus* was not economically viable, for example because they may have little suitable habitat or because a different species of coral trout such as *P. maculatus* or *P. laevis* may have predominated. Commercial catch data was not always recorded to a fine enough spatial scale to allow accurate allocation into Subbioregions, but the records were adequate to decide which Subbioregions should be left in the model and which should be omitted. The catch data used as input to the model itself were spatially classified only to Subregion level (see section 6.3 below).

The entire Capricorn–Bunker Region was omitted from the stock assessment because the catch of coral trout there was small, especially in recent years, and fishers there obviously did not strongly target coral trout. Also the underwater visual surveys conducted by AIMS showed dramatically different trends in coral trout abundance in the Capricorn–Bunker Region from the rest of the GBR (Figure 26, page 76). It appeared that the Capricorn–Bunker

Region had quite different recruitment dynamics from the rest of the GBR, and modelling this region would require many more parameters (recruitment deviations specific to the Capricorn–Bunker Region) to be added to the population model. We note that the Capricorn Bunker Region is physically separated from the Swains reefs by the deep Capricorn Channel.

The model also omitted potential inter-reef habitat for coral trout, and indeed all habitat deeper than 30 m, because this habitat is currently impossible to quantify. It is certainly the case that some coral trout live deeper than 30 m, and some live in areas not attached to particular coral reefs, but underwater video surveys show that most of the area between reefs is not suitable habitat for coral trout (Michael Cappo, AIMS, personal communication).

The Regions, Subregions, Bioregions and Subbioregions used in the model are listed in full in Table 24. Habitat areas of the Subbioregions are listed in Table 25.

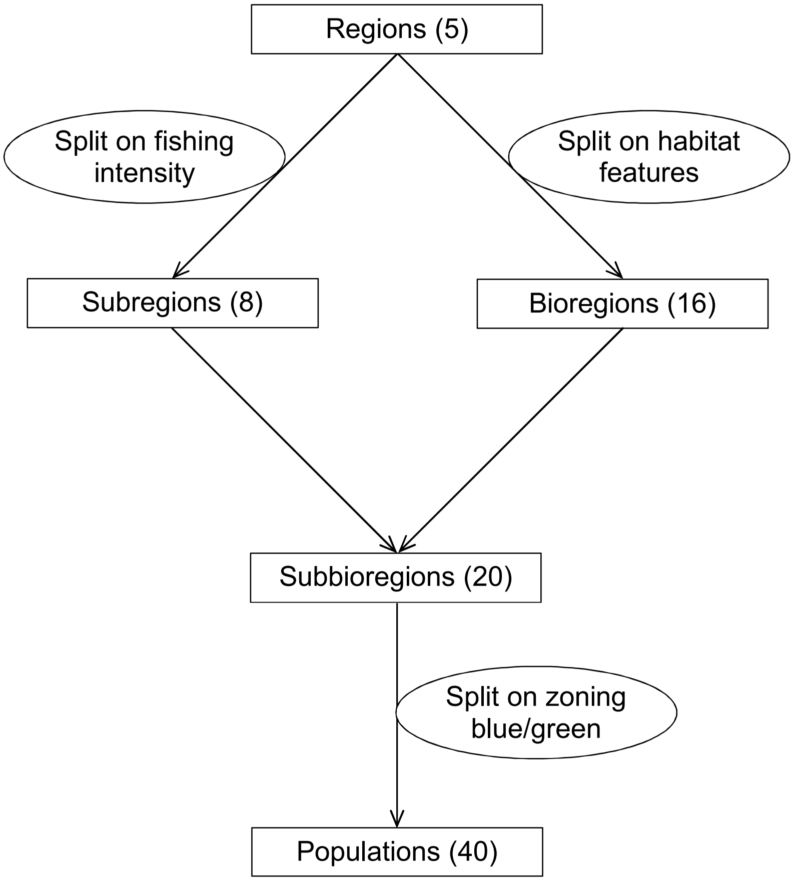


Figure 28: Regional structure of the coral trout population model, showing separate splits of Regions into Subregions and Bioregions, both of which then split into Subbioregions. Each Subbioregion contains two populations of coral trout, one in blue zones (open to fishing) and one in green zones (closed to fishing). The total number of each type of structural element is shown in parentheses. The model included only Subbioregions with substantial commercial catches of coral trout.

Table 24: Regions, Subregions, Bioregions and Subbioregions used in the model. Bioregions are shown in map form in Figure 6 on page 14, and Regions and Subregions in Figure 27. Subregions are listed from north to south, and Subbioregions are listed from west to east (inner shelf to outer shelf) within each Subregion (except in the Cape York Subregion where they both have the same shelf position and are listed from north to south).

Subbioregion	Bioregion & description		Subregion	Region
RC1	RC1	Mid shelf	Cape York	Far Northern
RC2 North	RC2	Protected mid shelf	Cape York	Far Northern
RD Central	RD	Open lagoon reefs	Lockhart River	Far Northern
RC2 Central	RC2	Protected mid shelf	Lockhart River	Far Northern
RB1 Central	RB1	Outer mid shelf	Lockhart River	Far Northern
RC2 South	RC2	Protected mid shelf	Pr. Charlotte Bay	Far Northern
RF1 North	RF1	Open lagoon reefs	Cooktown	Cooktown
RG1	RG1	Sheltered mid shelf	Cooktown	Cooktown
RA2 South	RA2	Outer barrier reefs	Cooktown	Cooktown
RG2 North	RG2	Exposed mid shelf	Cairns	Cairns–Townsville
RA3 North	RA3	Outer shelf	Cairns	Cairns–Townsville
RG2 South	RG2	Exposed mid shelf	Townsville	Cairns–Townsville
RA3 South	RA3	Outer shelf	Townsville	Cairns–Townsville
RK	RK	Strong tidal inner shelf	Mackay	Mackay
RHW	RHW	Strong tidal mid shelf	Mackay	Mackay
RHL	RHL	Hard Line	Mackay	Mackay
RA4	RA4	Strong tidal outer shelf	Mackay	Mackay
RHE	RHE	Strong tidal mid shelf	Swains	Swains
RSW-M	RSW-M	Swains mid	Swains	Swains
RSW-O	RSW-O	Swains outer	Swains	Swains

Table 25: Habitat areas for each Subbioregion, scaled to equivalent reef-slope area by the methods described in Chapter 5. Zoning is current from 1 July 2004.

Subbioregion	Blue-zone habitat (ha)	Green-zone habitat (ha)
RC1	3065	1195
RC2 North	8036	15210
RD Central	14702	4203
RC2 Central	15580	4115
RB1 Central	19440	2138
RC2 South	13446	8050
RF1 North	2656	1819
RG1	24680	7518
RA2 South	8699	7318
RG2 North	19919	5884
RA3 North	2550	2209
RG2 South	31850	8377
RA3 South	1691	1932
RK	18877	4880
RHW	41011	14294
RHL	35119	9864
RA4	6004	2286
RHE	7282	2987
RSW-M	14865	5918
RSW-O	3229	2029

6.3 Basic population dynamics

The model operated on calendar years, which were thought to better suit the biology of coral trout which spawns late in the year. Calendar years also matched the software ELFSim (Little et al., 2007) which was used in a parallel project to this stock assessment, funded by the Australian Government's Fisheries Research and Development Corporation (FRDC, project no. 2011/030), primarily to evaluate monitoring and management procedures for the fishery. Fishery quota, on the other hand, operates on Australian financial years, July to June.

Numbers of fish N present in the model at the beginning of a year were indexed by Population (k), year (t) and age (a). As proposed in section 1.5.3, sexes were not distinguished. Each Subbioregion contained two Populations, one zoned blue (open to fishing) and the other green (closed to fishing). The number of fish of age zero was set equal to the recruitment $R_{k\,t}$ to Population k in year t:

$$N_{k\,t\,0} = R_{k\,t}\,. \tag{6.1}$$

Recruitment is discussed in section 6.4 below.

For ages one year and upwards, population numbers are derived from those for the same year-class in the previous year (year $t-1$ and age $a-1$): for $1 \le a < a_{\max}$,

$$N_{k\,t\,a} = N_{k\,t-1\,a-1}\, e^{-M}\,(1 - V_{a-1}\,U_{k\,t-1}), \tag{6.2}$$

where $a_{\max}$ is the age of the oldest age-class in the model, M is the instantaneous natural mortality rate, V_a is the vulnerability to fishing at age a, and $U_{k\,t}$ is the harvest rate of population k in year t. The quantities $a_{\max}$, V_a and $U_{k\,t}$ are discussed in the following paragraphs.

The oldest age-class $a_{\max}$ was a "plus group", holding all fish of age $a_{\max}$ or older. The formula for it was slightly different to (6.2): for $a = a_{\max}$,

$$N_{k\,t\,a} = N_{k\,t-1\,a-1}\, e^{-M}\,(1 - V_{a-1}\,U_{k\,t-1}) + N_{k\,t-1\,a}\, e^{-M}\,(1 - V_a\,U_{k\,t-1}). \tag{6.3}$$

In fact $a_{\max}$ was chosen to be one year older than the oldest observed fish, i.e., $a_{\max} = 20$ yr (see section 1.5.3), so that all observed age frequencies were zero at age $a_{\max}$. This approach used all the information present in the age frequency data, so that no information was lost in truncating the age distribution at $a_{\max}$.

The fishery was assumed to start from the virgin (never fished) state in year 1, which was calendar year 1962, the first year in which the Queensland Fish Board recorded catch of coral trout. The level of fishing before then was assumed to be negligible. The population structure in year 1 was given by, for $1 \le a < a_{\max}$,

$$N_{k\,1\,a} = R_{k\,0}\, e^{-aM}\,,$$

where $R_{k\,0}$ is the deterministic number of recruits to population k in the virgin state (see section 6.4). For the plus group the formula took account of older fish: for $a = a_{\max}$,

$$N_{k\,1\,a} = R_{k\,0}\, e^{-aM} \big/ (1 - e^{-M}).$$

The vulnerability V_a is estimated in the model and represents the relative chance that a fish of age a that is present in the population will be caught by fishing or other sampling. Very small fish will not be caught even if they are in the vicinity, so have low vulnerability. Young fish will also be assigned low vulnerability if they are not in the vicinity, for example if they have a life cycle whereby young fish inhabit only very deep water that is not fished. This latter feature distinguishes the term "vulnerability" from the equipment-specific term "selectivity", which refers only to the capability of the fishing gear to catch fish that are at the location being fished. Young coral trout are not thought to inhabit different localities to older fish, except that they may be more inclined to take cover inside coral, so for coral trout it makes no difference whether the term "vulnerability" or "selectivity" is used. Other reef fish such as red throat emperor (*Lethrinus miniatus*) may reside elsewhere when young, so the term "vulnerability" is preferred for them (Leigh et al., 2006).

The model used a logistic function for vulnerability as a function of length. This function gradually increases from very low vulnerability for small fish, to approach 1 for large fish:

$$V_L^* = 1\big/\left[1 + \exp\{-(\log 19)\,(L - L_{50})/(L_{95} - L_{50})\}\right], \tag{6.4}$$

where L_{50} is the fork length at 50% vulnerability and L_{95} is the fork length at 95% vulnerability (see Haddon, 2001, p. 353); both L_{50} and the parameter $L_{95\,\mathrm{diff}} = L_{95} - L_{50}$ were estimated in the model. The asterisk distinguishes length-dependent vulnerability V_L^* from age-dependent vulnerability V_a. The conversion factor of 0.9409 was used to convert total length to fork length (see section 1.5.3, page 25).

Length-dependent vulnerability was converted to age-dependent vulnerability by Cabezon using the distribution of length at age in the middle of the year. This distribution was assumed to be normal, with mean given by the growth curve and standard deviation by the estimated coefficient of variation (see section 1.5.3): at a given age *a*, it produced the proportion of fish $p(L\,|\,a)$ in each length-class *L*, such that $\sum_L p(L\,|\,a) = 1$. Then the age-dependent vulnerability was given by

$$V_a = \sum_L p(L\,|\,a)\, V_L^*. \tag{6.5}$$

The model used 1 cm length categories with midpoints ranging from 1 cm to 70 cm, and calculated the vulnerability in the middle of the year, at exact age $a + \frac{1}{2}$.

The harvest rate $U_{k\,t}$ is the proportion of vulnerable fish in Population *k* that are caught in year *t*. In fact, catch sizes were specified only to Subregion level, so it depended only on the Subregion *g* that contained Population *k*:

$$U_{k\,t} = U_{g\,t}^*,$$

and the Subregion harvest rate $U_{g\,t}^*$ was calculated as the ratio of catch weight from Subregion *g* in year *t*, to the mid-year vulnerable biomass in Subregion *g* just before the start of the fishing pulse (which is described in section 6.1):

$$U_{g\,t}^* = C_{g\,t}\bigg/\sum_{k\in K(g)}\sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_a, \tag{6.6}$$

where W_a is the average mid-year weight of a fish of age *a*, and *K*(*g*) is the set of Populations that make up Subregion *g*. Treatment of green-zoned Populations is described in section 6.5.

Calculation of catch size when it had been recorded only to coarse (30-minute) resolution has been described in section 4.2.

Formulae (6.2) and (6.3) were used unchanged from Cabezon, and are appropriate when either the fishing intensity is low to moderate, or the non-vulnerable fish are absent from the fishing grounds. If the fishing intensity is very high then the vulnerability should more properly be applied to the fishing mortality rate than the harvest rate, which leads to a power relationship: (6.2) and (6.3) become

$$N_{k\,t\,a} = N_{k\,t-1\,a-1}\, e^{-M}\,(1 - U_{k\,t-1})^{V_{a-1}}$$

and

$$N_{k\,t\,a} = N_{k\,t-1\,a-1}\, e^{-M}\,(1 - U_{k\,t-1})^{V_{a-1}} + N_{k\,t-1\,a}\, e^{-M}\,(1 - U_{k\,t-1})^{V_a}.$$

The equation relating harvest rate to catch size also becomes much more complicated than (6.6). The fishing intensity on the GBR was not thought to be high enough to require this change. Therefore we used equations (6.2) and (6.3). A case of a fishery that would need the power relationship, together with the methodology for post-release mortality discussed in section 6.7, is that of a heavily fished catch-and-release fishery in which a typical fish may be caught and released several times in a single year; in Queensland, dusky flathead (*Platycephalus fuscus*) may be such a case.

Adjustments to the harvest rates to take account of green zones and minimum legal size limits are described in sections 6.5 and 6.7 respectively.

It can be seen from equation (6.6) that the population model was driven by catch size, not by fishing effort. It is possible to use fishing effort instead; for the commercial fishery the effort may be calculated by dividing the catch size by the standardised catch-rate from Chapter 4. Then the catch size becomes an observation that the model has to closely match, instead of being a direct input to the model. The effort-driven formulation makes the model more complex and inserts an extra parameter (the "catchability"), which scales fishing effort to an instantaneous fishing mortality rate.

An advantage of the effort-driven formulation is that it provides a cleaner and more logical way to extrapolate fishery data backward in time, before accurate records were kept. Fishing effort can be extrapolated backwards under reasonable assumptions about the growth of participation in the fishery, and, importantly, this is independent of trends in the abundance of fish. In a catch-driven formulation, extrapolation of catch size to use as input to the model is confounded by the trend in abundance of fish, which should properly be an output of the model, not an input.

For the coral trout fishery we judged that an effort-driven model did not provide sufficient benefit to justify the extra complexity, because

- The commercial coral trout fishery was small before the Queensland Fish Board began to keep records in 1962–63; and
- The recreational fishery is much smaller than the commercial fishery, meaning that there was little benefit in more accurately extrapolating the recreational catch size backward in time.

This model does not use Cabezon's multiple-fleet capability, whereby vulnerability can depend on both age and fleet in equations (6.2) and (6.3), and a separate harvest rate is defined for each fleet in a given year. In principle it may be desirable to allow different vulnerability functions for the commercial and recreational fleets, because commercial fishers target fish up to 1.5 kg, although they retain larger fish if they catch them, whereas recreational fishers especially value large fish but still value and retain legal-sized smaller fish. In practice, however, the only data from which to estimate vulnerability functions came from scientific surveys and were not specific to either the commercial or recreational fleet. Therefore it was not possible to distinguish fleet-specific vulnerability functions.

The capability for different vulnerability functions was, however, used for the underwater visual survey (UVS) data, which recorded the estimated length of each fish sighted by the divers. UVS did not involve fishing gear so was expected to have a different vulnerability (visibility) function to samples collected by fishing.

Targeting of medium-sized fish by commercial fishers provides some incentive to use what is known as a "dome-shaped" vulnerability function. Such a function peaks at a moderate size and then decreases for large fish, instead of continuing to increase like the logistic function.

We did not use dome-shaped vulnerability because there was insufficient evidence for it. Fishers retain large fish when they catch them, and it is unknown whether commercial fishers are able to choose fishing locations that are frequented by dinner-plate sized fish but not by large fish. Also, dome-shaped vulnerability can be dangerous because it postulates a large bank of spawning fish that are never observed. It is not known definitely whether these unobserved fish actually exist; if not, the spawning stock size could be grossly overestimated.

6.4 Recruitment

Spawning and recruitment were assumed to take place simultaneously at the beginning of each calendar year. The model allowed no time lag between spawning and subsequent recruitment. This formulation matched that used by both Cabezon and ELFSim, but differed from the standard theory used in previous stock assessments of other fisheries by DAFF (see, e.g., Haddon, 2001). Previous DAFF assessments have assumed spawning in the middle of the year, and subsequent recruitment at the beginning of the following year. In the Cabezon–ELFSim formulation, one year has to be added to the age of the fish, and the fecundity is taken at the beginning of the year, not the middle of the year. Either approach is adequate for long-lived fish; we preferred the Cabezon–ELFSim one only for compatibility with ELFSim.

There is debate over the distance that coral trout larvae migrate from the location where they were spawned, but current evidence favours short distances that are still sufficient for green zones (marine protected areas) to seed recruits into blue zones (see discussion in section 1.5.1). Therefore the model summed egg production over Bioregions, not large-scale elements such as Regions, or very small-scale ones such as Populations.

In a Population k that is contained in a Bioregion b, the recruitment $R_{k\,t}$ in year t followed a Beverton-Holt stock-recruitment relationship (Beverton and Holt, 1957) with random, annual lognormal deviations:

$$R_{k\,t} \big/ R_{k\,0} = e^{d_t} \, \frac{r\, S_{b\,t} \big/ S_{b\,0}}{1 + (r-1)\, S_{b\,t} \big/ S_{b\,0}},$$

where $S_{b\,t}$ is the egg production in year t in Bioregion b (blue zones and green zones combined), $R_{k\,0}$ and $S_{b\,0}$ are the deterministic values of $R_{k\,t}$ and $S_{b\,t}$ in a virgin (never fished) population, $r > 1$ is the recruitment compensation ratio (see section 1.7), and d_t is the log-recruitment deviation.

The egg production in Bioregion b, which comprises a set of Populations $K_B(b)$, is

$$S_{bt} = \sum_{k \in K_B(b)} \sum_{a=1}^{a_{\max}} x_a \, N_{k\,t\,a} \,,$$

where x_a is the product of the maturity proportion and the fecundity at age a (see section 1.5.3).

The recruitment compensation ratio r was estimated in the model and was common to all Regions.

Within each Subbioregion, the parameters $R_{k\,0}$ were made proportional to the habitat areas H_k of the Populations. A value of the recruitment density $R_{k0} \big/ H_k$, as a number of recruits per hectare, was estimated within the model for each Subbioregion; the same density value $R_{k0} \big/ H_k$ was used for both blue zones and green zones.

The log-recruitment deviations d_t were estimated within the model and followed a normal distribution with mean zero. A lower bound of 0.1 was applied to the standard deviation to prevent the likelihood from becoming infinite. Cabezon applies a bias-correction factor so that the expected value of e^{d_t} is equal to 1. We did not apply this, as we preferred to set the median equal to 1 rather than the arithmetic mean equal to 1.

There was only one recruitment deviation per year, covering all Regions, because Region-specific deviations could not be estimated reliably from the available data. We note that, judging from the UVS data collected by AIMS, GBR-wide recruitment deviations would not have fitted the Capricorn–Bunker Region and, if the catch sizes from this Region had been large enough to justify including it in the model, an extra sequence of recruitment deviations would have been needed just for this Region (see section 6.2).

6.5 Green zone fishing parameter

Fishing in green zones was handled by a parameter f_{green} which acted as a scaling factor on the vulnerable biomass: a proportion f_{green} of the vulnerable biomass in equation (6.6) was considered vulnerable if the Population was zoned green, where $0 \le f_{\text{green}} \le 1$.

For a Subregion g comprising a set of blue-zoned Populations $K_{\text{blue}}(g)$ and a set of green-zoned Populations $K_{\text{green}}(g)$, the vulnerable biomass in Population k is equal to

$$\begin{cases} \sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_a\,, & \text{if } k \in K_{\text{blue}}(g) \\ f_{\text{green}} \sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_a\,, & \text{if } k \in K_{\text{green}}(g) \end{cases}$$

and equation (6.6) becomes

$$U^{*}_{g\,t} = C_{g\,t} \Bigg/ \left\{ \sum_{k \in K_{\text{blue}}(g)} \sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_a + f_{\text{green}} \sum_{k \in K_{\text{green}}(g)} \sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_a \right\}. \quad (6.7)$$

When Population k is zoned green, the population dynamic equations (6.2) and (6.3) become respectively

$$N_{k\,t\,a} = N_{k\,t-1\,a-1}\, e^{-M} (1 - f_{\text{green}}\, V_{a-1}\, U_{k\;t-1}),$$

and

$$N_{k\,t\,a} = N_{k\,t-1\,a-1}\, e^{-M} (1 - f_{\text{green}}\, V_{a-1}\, U_{k\;t-1}) + N_{k\,t-1\,a}\, e^{-M} (1 - f_{\text{green}}\, V_{a}\, U_{k\;t-1}).$$

There were no data from which f_{green} could be estimated reliably. Therefore it was fixed to the value 0.2; i.e., the harvest rate in green zones was assumed to be 20% of that in neighbouring blue zones, based on advice from industry and government. It was clear that there was substantial fishing in green zones, but many fishers did not indulge in it, and those that did would have had to put time into avoiding being caught, which must have made their fishing less effective. We regarded the figure of 20% as reasonable.

The vulnerable biomass in subregion g at the beginning of year t is equal to the denominator in equation (6.7). To use vulnerable biomass as an abundance index to compare to catch rates, we adjust it to the middle of the fishing pulse:

$$B_{g\,t} = \sqrt{1 - U^{*}_{g\,t}} \left\{ \sum_{k \in K_{\text{blue}}(g)} \sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_a + f_{\text{green}} \sum_{k \in K_{\text{green}}(g)} \sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_a \right\}, \quad (6.8)$$

This is different to, and slightly more accurate than, the equation used in Cabezon, which uses $1 - \tfrac{1}{2} U^{*}_{g\,t}$ in place of the square-root factor. The difference was expected to be negligible, but equation (6.8) is more logical when the social learning parameter is applied in section 6.8.

6.6 Changes in zoning

Zoning of reefs changed from time to time (see Table 1, page 19). The biggest change in zoning came in July 2004 when the proportion of the GBR that was closed to fishing increased from about 5% to 33%. This change meant that fishers had a smaller area in which to legally fish, and had to fish it more intensively, which would have resulted in a decrease in catch rates. It was considered desirable for the model to capture this effect.

Habitat area, denoted H_k in section 6.4, is now indexed also by year (t), and denoted $H_{k\,t}$. Formally, $H_{k\,t}$ denotes the average habitat area of Population k in year t.

Suppose that zoning changed in year t in a Subbioregion comprising Populations k (zoned blue) and k^{*} (zoned green). It is assumed that all the rezoning in the Subbioregion in year t is in the same direction, either all from blue to green or all from green to blue. The projected population numbers under the previous year's zoning are given by the right-hand sides of equations (6.1), (6.2) and (6.3), and are denoted $N_{k\,t\,a}^{\text{proj}}$. If the zoning change is from blue to green (the more common case), then $H_{k\,t} < H_{k\,t-1}$ and $H_{k^*\,t} > H_{k^*\,t-1}$. The total habitat in the Subbioregion is still the same, i.e., $H_{k\,t} + H_{k^*\,t} = H_{k\,t-1} + H_{k^*\,t-1}$. The population numbers are adjusted by the formulae

$$N_{k\,t\,a} = \left(H_{k\,t} \big/ H_{k\,t-1}\right) N_{k\,t\,a}^{\text{proj}}$$

and

$$N_{k^*t\,a} = N_{k^*t\,a}^{\text{proj}} + \left\{\left(H_{k\,t-1} - H_{k\,t}\right) \big/ H_{k\,t-1}\right\} N_{k\,t\,a}^{\text{proj}}.$$

If the zoning change is from green to blue (the rarer case), the formulae are

$$N_{k\,t\,a} = N_{k\,t\,a}^{\text{proj}} + \left\{\left(H_{k\,t} - H_{k\,t-1}\right) \big/ H_{k^*\,t-1}\right\} N_{k^*t\,a}^{\text{proj}}$$

and

$$N_{k^*t\,a} = \left(H_{k^*t} \big/ H_{k^*\,t-1}\right) N_{k\,t\,a}^{\text{proj}}.$$

Accounting for zoning changes in this way also means that vulnerable biomasses can no longer be used on their own as abundance indices, because they change with zoning in ways that are unrelated to abundance. Therefore the vulnerable biomass $B_{g\,t}$ in (6.8) has to be scaled by the "vulnerable habitat area"

$$\widetilde{H}_{g\,t} = \sum_{k \in K_{\text{blue}}(g)} H_{k\,t} + f_{\text{green}} \sum_{k \in K_{\text{green}}(g)} H_{k\,t} \tag{6.9}$$

to produce an abundance index

$$B_{g\,t} \big/ \widetilde{H}_{g\,t}$$

that is comparable from year to year.

6.7 Size limits

Minimum legal sizes (MLS), which could change over time, were handled by adjusting the vulnerability function in equation (6.5), and specifying a post-release mortality rate u (the appropriate value of which is discussed in section 1.5.1). Let the MLS be L_{MLS}. Then (6.5) is altered to

$$V_a = u \sum_{L < L_{\text{MLS}}} p(L \mid a)\, V_L^* + \sum_{L \ge L_{\text{MLS}}} p(L \mid a)\, V_L^*,$$

which is used in the population dynamic equations (6.2) and (6.3).

For the harvest-rates and abundance indices defined by equations (6.6) and (6.8), the catch and catch-rate are assumed to comprise only legal-sized fish. Therefore we define a separate vulnerability function for fish that the fishers keep,

$$V_{a\,\text{keep}} = \sum_{L \ge L_{\text{MLS}}} p(L \mid a)\, V_L^*,$$

and (6.6) becomes

$$U_{g\,t}^* = C_{g\,t} \Bigg/ \sum_{k \in K(g)} \sum_{a=0}^{a_{\max}} N_{k\,t\,a}\, e^{-M/2}\, W_a\, V_{a\,\text{keep}}.$$

For green-zone fishing and social learning (see below), V_a is also replaced by $V_{a\,\text{keep}}$ in (6.7) and (6.10).

An ideal treatment of a MLS would also involve increasing the weight-at-age of fish that were caught, and decreasing the weight-at-age of the remaining fish that were not caught. This would have imposed a substantial programming and computational overhead for little perceived benefit, and was not pursued in this assessment.

6.8 Social learning parameter

Social learning (hook shyness) by coral trout was handled by including a parallel or "shadow" population of fish with a higher fishing mortality rate than the actual population, which is intended to depress the catch rates when the population learns not to take bait as a result of being fished.

In the presence of social learning, the square-root term in (6.8) is raised to the power γ, the social learning parameter, to produce the shadow vulnerable biomass

$$\widetilde{B}_{gt} = \left(1 - U^*_{gt}\right)^{\gamma/2} \left\{ \sum_{k \in K_{\text{blue}}(g)} \sum_{a=0}^{a_{\max}} N_{kta}\, e^{-M/2}\, W_a V_a + f_{\text{green}} \sum_{k \in K_{\text{green}}(g)} \sum_{a=0}^{a_{\max}} N_{kta}\, e^{-M/2}\, W_a V_a \right\}. \quad (6.10)$$

The shadow biomass $\widetilde{B}_{gt}$ was used in abundance indices, in place of the true vulnerable biomass B_{gt}, to match to the standardised commercial catch rates calculated in Chapter 4. The formulation that avoids the power relationship in (6.10) uses a factor of $1-(\gamma/2)U^*_{gt}$ instead of the factor involving the power $\gamma/2$. This is undesirable because it can easily produce negative values (equivalent to taking a catch greater than the available biomass). Admittedly, square-roots and power relationships can cause trouble in the automatic differentiation routines used in ADMB, because the derivative becomes infinite when the argument is zero and the power is less than 1. They were judged to be necessary here, despite the potential problems.

The social learning parameter γ was estimated in the model, with the restriction only that it had to be greater than zero. In the absence of social learning it would have the value 1. If social learning is present its value should be greater than 1. For the results reported here, the parameter was taken to be 1. This is because the majority harvest is taken from Cooktown south where social learning was assumed to have already taken place, moving the catch rate depletion relationship to the proportional region of Figure 12.

6.9 List of model parameters

The parameters used in the model are listed in Table 26. The recruitment deviations d_t were constrained to sum to zero so that their mean was not confounded with the recruitment-density parameters $R_{k0\,\text{dens}}$. There were no age-frequency data from which to estimate any recruitment deviations before the 1981 year class or after the 2007 year class; the most recent sample was from spring 2009. Therefore the recruitment deviations were fixed at zero (deterministic recruitment) for year classes outside the range 1981–2007. Catch rates and abundance are not well correlated for coral trout, so catch rates were not considered adequate for estimation of recruitment deviations from catch-rate data alone.

6.10 Data and likelihoods

6.10.1 Data

A list of all data used in the assessment has been provided in section 1.8. The data actually used in the model are listed in Table 27: mostly these data were not raw data but had been derived from the raw data by methods presented in previous chapters.

The data listed above the bold line in Table 27 were used in the model's internal calculations. The data below the line were used to match the model's predictions, as described in the following sections. The coefficients of variation (CVs, ratios of standard error to the mean value) of the abundance data were the standard errors of log-transformed parameters in generalised linear models. These CVs included only observation error (error that can be made arbitrarily small by collecting more data) and not process error (error caused by lack of fit of the model, which generally is not reduced by collecting more data). Therefore, to account for possible process error, they were used in the model only as lower bounds for the CVs.

Table 26: Parameters used in the model. The Length column is the number of degrees of freedom in the parameter. The Value is listed when it is fixed, and left blank when estimated.

Name	Length	Value	Description
$R_{k\,0\,\mathrm{dens}}$	16		Virgin density of recruits, R_{k0}/H_k (number of recruits of age 0 per hectare), by Bioregion (k = 1, …, 16)
r	1		Recruitment compensation ratio
d_t	26		Recruitment deviations (years 1981, …, 2007), constrained to sum to zero; lower bound of 0.1 on standard deviation
M	1		Instantaneous natural mortality rate
L_{50}	1		Fork length at 50% vulnerability to fishing
$L_{95\,\mathrm{diff}}$	1		Fork length at 95% vulnerability to fishing, minus L_{50}
L_{50}^{UVS}	1		Fork length at 50% vulnerability to UVS
$L_{95\,\mathrm{diff}}^{\mathrm{UVS}}$	1		Fork length at 95% vulnerability to fishing, minus L_{50}^{UVS}
f_{green}	1	0.20	Intensity of fishing in green zones, as a fraction of that in neighbouring blue zones
u	1	0.25	Discard mortality rate
γ	1		Social learning parameter

Table 27: Data used in the model. The data listed above the bold line were used in the model's internal calculations, while those below the line were used to match the model's predictions.

Name	Description
L_∞, K, t_0	Von Bertalanffy growth curve parameters (see section 1.5.3)
$\mathrm{CVL}_{\mathrm{min}}$	Coefficient of variation of length about the mean at age 1 (see section 1.5.3)
$\mathrm{CVL}_{\mathrm{max}}$	Coefficient of variation of length about the mean at age $a_{\mathrm{max}} - 1$ (see section 1.5.3)
W_a	Average mid-year weight of a fish of age a
x_a	Product of maturity proportion and fecundity at age a (see section 1.5.3)
MLS_t	Minimum legal size (fork length) by year
$H_{k\,t}$	Habitat area (hectares) by Subbioregion and year (see Chapter 5 and section 6.2)
$C_{g\,t}$	Catch size by Subregion and year, sum of commercial and recreational catches, interpolated and extrapolated backwards in time where necessary (see Chapter 4)
$Y_{g\,t}$	Relative abundances from standardised commercial catch rates by Subregion and year (see Chapter 4)
$\mathrm{CVY}_{g\,t}$	Coefficients of variation of $Y_{g\,t}$, used as lower bounds in the model
$A_{k\,t}$	Absolute abundances (number of fish per hectare of reef slope) from underwater visual surveys (UVS) contracted by GBRMPA and the ELF Project, by Subbioregion and year (see Chapter 5 for discussion of scaling)
$\mathrm{CVA}_{g\,t}$	Coefficients of variation of $A_{g\,t}$, used as lower bounds in the model
$Y_{k\,t}^{\mathrm{UVS}}$	Relative abundances from UVS conducted by AIMS and Fisheries Queensland, by Subbioregion and year
$\mathrm{CVY}_{k\,t}^{\mathrm{UVS}}$	Coefficients of variation of $Y_{k\,t}^{\mathrm{UVS}}$, used as lower bounds in the model
$y_{k\,t\,a}$	Age frequencies collected by JCU (Russ et al., 1995), the ELF Project and Fisheries Queensland (see section 1.5.2), by Subbioregion, year and age
$y_{k\,t\,\ell}^{\mathrm{UVS}}$	Length frequencies from UVS contracted by GBRMPA and the ELF Project, by Subbioregion, year and length class (the same surveys that produced $A_{k\,t}$)

6.10.2 Likelihood for relative abundance measures

A relative abundance index $Y_{g\,t}$ follows a lognormal distribution. The abundance from standardised catch rates is assumed to be proportional to the social-learning-adjusted vulnerable biomass $\tilde{B}_{gt}$ from equation (6.10), scaled by the corresponding habitat area $\tilde{H}_{gt}$ from equation (6.9). The constant of proportionality is captured in the parameter μ below: it accounts for the fact that catch rates (numbers of fish caught by a line fisher per dory-day of fishing) measure only the relative abundance of fish, and do not directly measure the number of fish per hectare of habitat. This parameter is not used when the number of fish per hectare is measured directly, as in underwater visual surveys (see section 6.10.3 below).

When the mean μ and standard deviation $\sigma_{g\,t}$ of $\log Y_{gt} - \log\left(\tilde{B}_{gt}/\tilde{H}_{gt}\right)$ are specified, the likelihood is

$$\prod_g \prod_t \left(\left\{ 1 \big/ \left(\sqrt{2\pi}\sigma_{gt} \right) \right\} \exp\left[-\tfrac{1}{2} \left\{ \log Y_{gt} - \log\left(\tilde{B}_{gt}/\tilde{H}_{gt}\right) - \mu \right\}^2 \big/ {\sigma_{gt}}^2 \right] \right),$$

where subscripts *g* and *t* denote Subregions and years respectively. It is convenient to use the negative log-likelihood (NLL), which, omitting the constant factors of $\sqrt{2\pi}$ above, is

$$\ell = \sum_g \sum_t \left[\log \sigma_{gt} + \tfrac{1}{2} \left\{ \log Y_{gt} - \log\left(\tilde{B}_{gt}/\tilde{H}_{gt}\right) - \mu \right\}^2 \big/ {\sigma_{gt}}^2 \right].$$

The standard deviation $\sigma_{g\,t}$ is set to $\mathrm{CVY}_{g\,t}$ (see Table 27) multiplied by a scale factor $\sigma \geq 1$ which is intended to account for process error (see section 6.10.1). Then the NLL, omitting constant terms, is

$$\ell = \sum_g \sum_t \left[\log \sigma - \tfrac{1}{2} \log w_{gt} + \tfrac{1}{2} w_{gt} \left\{ \log Y_{gt} - \log\left(\tilde{B}_{gt}/\tilde{H}_{gt}\right) - \mu \right\}^2 \big/ \sigma^2 \right], \tag{6.11}$$

where $w_{gt} = 1 \big/ \mathrm{CVY}_{gt}^2$.

Standard estimators of μ and σ^2 are:

$$\hat{\mu}_Y = \sum_g \sum_t w_{gt} \left\{ \log Y_{gt} - \log\left(\tilde{B}_{gt}/\tilde{H}_{gt}\right) \right\} \Big/ \sum_g \sum_t w_{gt}$$

and

$$\hat{\sigma}_Y^2 = \sum_g \sum_t \left[w_{gt} \left\{ \log Y_{gt} - \log\left(\tilde{B}_{gt}/\tilde{H}_{gt}\right) - \hat{\mu}_Y \right\}^2 \right] \big/ (n_Y - 1). \tag{6.12}$$

Substituting these expressions into (6.11) provides a likelihood that depends only on data ($Y_{g\,t}$, $w_{g\,t}$ and $\tilde{H}_{gt}$) and model predictions ($\tilde{B}_{gt}$):

$$\ell = (n_Y - 1) \log \tilde{\sigma}_Y + \tfrac{1}{2} (n_Y - 1) \hat{\sigma}_Y^2 / \tilde{\sigma}_Y^2, \tag{6.13}$$

where n_Y is the total number of Subregion–year combinations in the index series, and $\tilde{\sigma}_Y$ is the estimate of σ taking account of its lower bound $\sigma_{Y\min} = 1$:

$$\tilde{\sigma}_Y = \max(\hat{\sigma}_Y, \sigma_{Y\min}). \tag{6.14}$$

Formula (6.13) is similar to the negative log-likelihood derived by Haddon (2001, p. 89) but includes the adjustment term for the lower bound on σ.

The "max" function is not suitable for ADMB because its derivative is discontinuous. In fact, it is better not to calculate $\hat{\sigma}_Y$ either, but to use $\hat{\sigma}_Y^2$ directly from (6.12), because $\hat{\sigma}_Y$ involves a square root which causes trouble if $\hat{\sigma}_Y^2 = 0$. Therefore we used the following expression for $\tilde{\sigma}_Y$:

$$\tilde{\sigma}_Y = \sqrt{\tfrac{1}{2}(\hat{\sigma}_Y^2 + \sigma_{Y\min}^2) + \sqrt{\tfrac{1}{4}(\hat{\sigma}_Y^2 - \sigma_{Y\min}^2)^2 + 4\delta^2 \sigma_{Y\min}^4}}\,, \tag{6.15}$$

where $\delta > 0$ is a smoothness parameter that took the value 0.1. The value $\delta = 0$ makes (6.15) the same as (6.14), which is the formula that has to be avoided. The smoothing has the side effect of shifting the value of $\tilde{\sigma}_Y$ at $\hat{\sigma}_Y = \sigma_{Y\min}$ up to approximately $(1 + \delta)\, \sigma_{Y\min}$ instead of the desired value of $\sigma_{Y\min}$. The value $\delta = 0.1$ shifted it up 10%, which was held to be a reasonable compromise.

For UVS data, the relative abundance index Y_{kt}^{UVS} uses numbers instead of biomass, and does not have to be adjusted for social learning. Also UVS data are defined on Populations instead of Bioregions, because the reefs on which the UVS data were collected are known.

UVS was conducted all through the year, but the ELF Project surveys took place in the spring, which in the model is pushed forward to the beginning of the following calendar year. Therefore the UVS data were assumed to be collected at the beginning of the year, and the dates of surveys were rounded to either beginning of the calendar year in which they were conducted, or the beginning of the following year.

Instead of the adjusted biomass $\tilde{B}_{gt}$ in a Subregion g, UVS uses the total number of fish vulnerable (i.e., visible) to UVS in Population k:

$$\tilde{N}_{kt} = \sum_{a=0}^{a_{\max}} V_a^{\text{UVS}} N_{kta} ,$$

where V_a^{UVS} is the age-dependent vulnerability to UVS, which is defined in the same way as V_a (see equations (6.4) and (6.5)), but with different parameters (see Table 26). Instead of $\tilde{B}_{gt} \big/ \tilde{H}_{gt}$, the model's abundance index is now $\tilde{N}_{kt} \big/ H_{kt}$. The negative log-likelihood is the same as (6.13) but with a different value n_Y^{UVS} for n_Y, and different expressions $\hat{\sigma}_Y^{\text{UVS}}$ for $\hat{\sigma}_Y$ and $\tilde{\sigma}_Y^{\text{UVS}}$ for $\tilde{\sigma}_Y$. The lower bound $\sigma_{Y\min}^{\text{UVS}}$ still takes the value 1.

6.10.3 Likelihood for absolute abundance measures

A discussed above, the likelihoods for absolute abundance measures A_{kt} do not contain the mean-offset parameter μ in (6.11), as it is set equal to zero. Then (6.11) becomes

$$\ell = \sum_k \sum_t \left[\log\sigma - \tfrac{1}{2}\log w_{kt} + \tfrac{1}{2} w_{kt} \left\{ \log A_{kt} - \log\left(\tilde{N}_{kt} \big/ H_{kt} \right) \right\}^2 \big/ \sigma^2 \right],$$

where k denotes a Subbioregion and t a year; the standard deviation parameter σ and weighting factors w_{kt} are different to those in (6.11). The final negative log-likelihood (6.13) becomes

$$\ell = n_A \log\tilde{\sigma}_A + \tfrac{1}{2} n_A\, \hat{\sigma}_A^2 \big/ \tilde{\sigma}_A^2 , \tag{6.16}$$

where n_A is the total number of Subbioregion–year combinations in the index series,

$$\hat{\sigma}_A^2 = \sum_k \sum_t \left[w_{kt} \left\{ \log A_{kt} - \log\left(\tilde{N}_{kt} \big/ H_{kt} \right) \right\}^2 \right] \big/ n_A ,$$

$$\tilde{\sigma}_A = \max(\hat{\sigma}_A , \sigma_{A\min}) \tag{6.17}$$

and $\sigma_{A\min} = 1$. The number of degrees of freedom is n_A, not $n_A - 1$, because the mean μ is no longer estimated. The max function in (6.17) was also made into a smooth function in the same way as in equation (6.15).

6.10.4 Likelihood for age frequencies and length frequencies

An age frequency consists of a number of fish y_a measured in each age class $a = 0, \ldots, a_{\max}$ (see section 6.3 and Table 27). When each fish is considered to be independent of all other fish, the likelihood of a single age frequency is multinomial:

$$\binom{y_{\text{tot}}}{y_0, \ldots, y_{a_{\max}}} \prod_{a=0}^{a_{\max}} p_a^{\;y_a}, \tag{6.18}$$

where y_{tot} is the total number of fish whose ages are measured (sum of the y_a), p_a is the model's predicted proportion of fish in age-class a, the multinomial coefficient is defined as

$$\binom{y_{\text{tot}}}{y_0, \ldots, y_{a_{\max}}} = y_{\text{tot}}! \Big/ \prod_{a=0}^{a_{\max}} y_a!,$$

and the factorial function is defined as

$$y! = \prod_{i=1}^{y} i.$$

In practice, sampled fish are not independent, and instead of the total number y_{tot} the sample has an "effective sample size" that is usually much less than y_{tot} (Pennington and Vølstad, 1994; McAllister and Ianelli, 1997; Francis, 2011).

We deal with the problem of effective sample size by adjusting the multinomial likelihood. The approach estimates the effective sample size from the "raggedness" of the age-frequency distribution: a smooth distribution gives a high effective sample size, and a very ragged one gives a low effective sample size. It does not use the actual sample size y_{tot}.

We accept the point made by Francis (2011) that this approach can overestimate the effective sample size if the sample distribution is smooth but biased towards either old fish or young fish. We believe that this is not a major problem in fishery-independent sampling of Queensland fish populations, in which the sample age distributions tend to be ragged and show little sign of smoothness. The method proposed by Francis is extremely complex. We put substantial resources into trying to make it work in the eastern king prawn fishery on the Australian east coast, but without success. That project eventually used the same approach documented here (O'Neill et al., 2014).

We believe that the method we use, although not perfect, is the best method currently available for adjusting age-frequency likelihoods for effective sample size. It differs from the one used by Cabezon which abandoned the multinomial likelihood and replaced it by a sum of squares analogous to a chi-square statistic. We retain the multinomial likelihood as far as possible.

Firstly, we note that zero values of y_a in (6.18) make no contribution to the likelihood. Hence we restrict the likelihood to ages a for which $y_a > 0$. We let q denote the number of such ages and Q denote the set of these ages. Then the likelihood (6.18) becomes

$$\left\{ y_{\text{tot}}! \Big/ \prod_{a \in Q} y_a! \right\} \prod_{a \in Q} p_a^{\;y_a}. \tag{6.19}$$

We introduce the effective sample size, denoted T, so that an observation of y_a fish of age a in the sample of size y_{tot} is transformed to an effective observation of $(T/y_{\text{tot}})y_a$ fish from a sample of size T. We also treat the likelihood (6.19) like a probability density function (p.d.f.) of the y_a in $q-1$ dimensions; the number of dimensions is $q-1$ rather than q because the y_a are not independent but are constrained to sum to y_{tot}. The transformed likelihood has to remain a p.d.f. of y_a, not of $(T/y_{\text{tot}})y_a$, which necessitates multiplying by the factor $(T/y_{\text{tot}})^{q-1}$. Therefore the likelihood (6.19) is transformed to

$$(T/y_{\text{tot}})^{q-1} \left\{ T! \Big/ \prod_{a \in Q} (Ty_a/y_{\text{tot}})! \right\} \prod_{a \in Q} p_a^{\;Ty_a/y_{\text{tot}}}. \tag{6.20}$$

When Ty_a/y_{tot} is not an integer, the factorial function can be replaced by the gamma function, a mathematical special function which is defined for non-integer values and reproduces the factorial function at integer values.

We approximate the factorial function by Stirling's formula which is a well-known formula in mathematics:

$$x! \sim \sqrt{2\pi x}\, x^x e^{-x}\,.$$

This approximation becomes extremely close as $x \to \infty$, but for practical purposes is also close for small x, e.g., $x \geq 1$. Then, omitting constant factors and factors involving only the data y_a, the likelihood (6.20) becomes

$$T^{q-1}\left\{T^{T+\frac{1}{2}}e^{-T}\Big/\left[T^{q/2}\prod_{a\in Q}\left\{(Ty_a/y_{\text{tot}})^{Ty_a/y_{\text{tot}}}\,e^{-Ty_a/y_{\text{tot}}}\right\}\right]\right\}\prod_{a\in Q}p_a{}^{Ty_a/y_{\text{tot}}},$$

which, with some algebraic manipulation, can be simplified to

$$T^{(q-1)/2}\prod_{a\in Q}(p_a/\hat{p}_a)^{T\hat{p}_a},$$

where $\hat{p}_a = y_a/y_{\text{tot}}$ is the observed proportion of fish of age a in the sample. This produces the negative log-likelihood

$$\ell = -\tfrac{1}{2}(q-1)\log T + T\sum_{a\in Q}\hat{p}_a\log(\hat{p}_a/p_a)\,. \tag{6.21}$$

(Note that $p_a/\hat{p}_a$ has been replaced by $\hat{p}_a/p_a$ to reverse the sign of the log factor.)

The effective sample size T is estimated by maximum likelihood, by minimising the negative log-likelihood (6.21):

$$\hat{T} = \tfrac{1}{2}(q-1)\Big/\sum_{a\in Q}\hat{p}_a\log(\hat{p}_a/p_a)\,. \tag{6.22}$$

In the theory of generalised linear models (see McCullagh and Nelder, 1989, p. 197), this is also the estimate produced by equating the deviance of the multinomial model, $2T\sum\hat{p}_a\log(\hat{p}_a/p_a)$ to its asymptotic, large-sample expectation, $q-1$. Substituting the estimate (6.22) into the negative log-likelihood (6.21), and ignoring the resulting constant term, yields the final negative log-likelihood for the age-frequency sample:

$$\ell = -\tfrac{1}{2}(q-1)\log\hat{T}\,. \tag{6.23}$$

For every available age-frequency sample, the negative log-likelihood given by (6.23) and (6.22) is added into the overall negative log-likelihood for the model. Using this formulation it would be easy to impose a lower bound $T_{\min}$ on the effective sample size T for each sample, e.g., to force $T \geq 1$ or $T \geq 2$, but we did not consider it necessary to do that. The negative log-likelihood for such a case would be

$$-\tfrac{1}{2}(q-1)\log\tilde{T} + \tfrac{1}{2}(q-1)\tilde{T}/\hat{T},$$

where $\tilde{T} = \max(\hat{T}, T_{\min})$.

We note that, in the fishery-independent sampling programs that provided age-frequency data for this assessment, nearly all fish sampled were aged. Therefore we did not need to deal with the additional complexity of age-length keys to combine length frequencies with ageing data on some of the fish to produce overall age-frequencies.

Length-frequency samples were handled in exactly the same way as age-frequency samples. Each age-frequency or length-frequency produced a term of the form (6.23) that was added into overall negative log-likelihood for the model.

Because the age-frequency and length-frequency samples were collected scientifically and were not subject to minimum legal size limits, the adjustments to the vulnerability functions described in section 6.7 were not employed in calculating the predicted age- and length-frequencies.

6.10.5 Likelihood for recruitment deviations

The recruitment deviations d_t were assumed to follow a lognormal distribution and were treated identically to the relative abundance indices in section 6.10.2. This produced a single term to add into the overall negative log-likelihood.

7. Results from the population dynamic model

The population dynamic model has provided information about coral trout stocks and has highlighted conflicts between difference sources of input data. The major conflict is that

- Underwater visual survey data indicate a stable, although not extremely large, population size, whereas
- Commercial catch rates, on the other hand, decreased sharply from 2008 to 2011, consistent with a population that was falling dramatically due to the impact of fishing.

Past and recent underwater visual survey data establish that the population of coral trout could not have fallen as fast as the commercial catch rates between 2008 and 2011. Therefore the falls in catch rates must have been caused primarily by factors other than population size. Tropical cyclones are the obvious factor. There is substantial evidence that they have big effects on the fishery, but modelling of these effects has so far been elusive (see section 4.6.2).

Because of the conflict between input sources, the model inferred the coral trout stock to be less productive than expected, with a low estimate for the recruitment compensation ratio. This was the best way for it to balance a fairly high stock size and a moderate level of fishing with an apparent high impact of fishing on the catch rates.

The estimated exploitable biomass ratio for blue zones from the model in 2012 was 60% (B_{2012}/B_0). Stronger allowance for the effect of tropical cyclones may result in a higher estimate. The estimated value of the recruitment compensation ratio, r, was 3.99. This is lower than values that have been used for groupers in North America, which range between 12 and 25 (see section 1.7, page 27). All of the biological evidence about coral trout is that the population should be very capable of reproducing itself because the species is long-lived and matures early in life (see section 1.5).

More reasonable results were obtained by fixing the value of r to 10, which is still low by the standards of other grouper fisheries. This value produced a maximum sustainable yield (MSY) of 2010 tonnes per year, which was achieved at a harvest rate of about 69% and a ratio of exploitable biomass to virgin exploitable biomass of 19.4%. This harvest rate is much higher than would be practical for the GBR because of the low catch rates and fisher-profitability problems that would result. For comparison to this suggested level of MSY, the current total allowable commercial catch (TACC) is 1288 t, and the current recreational harvest in the offshore Subbioregions used in the stock assessment is slightly under 100 t, making a total of slightly under 1400 t. The actual commercial harvest in 2013 was about 850 t.

Equilibrium yield estimates are tabulated in Tables 28–31. Estimates are provided for two values of recruitment compensation ratio r ($r = 10$ and $r = 3.99$), two definitions of coral trout biomass (exploitable biomass for all legal-sized fish in GBR blue zones in the area of the commercial fishery, and total adult biomass for all legal sized fish in both green and blue zones in the Bioregions fished by the commercial fishery), and four target biomass levels: higher biomass ratios are associated with higher profitable catch rates. As noted above, more reasonable results were obtained by fixing the value of r to 10. This r value aligns more closely to the biology of coral trout and implies that the populations are better buffered against recruitment impacts of harvest than for $r = 3.99$. The underwater visual survey data also align more closely with the higher r value. Statistical confidence intervals (approximately 2 standard errors) on estimates were ± 22% and give an indication of the level of uncertainty surrounding the predictions. Interpretation of the yield estimates should be made cautiously, acknowledging the uncertainty caused by conflicts between different sources of input data.

The stock model (r estimated at 3.99) predicts commercial catch rates relatively well: an example fit for the Cairns Subregion is illustrated in Figure 29. Underwater visual survey data on the other hand are predicted relatively poorly; see Figures 30 and 31. Consequently, given our preference for the underwater visual survey data over the commercial catch rate data, we urge caution in the interpretation of these results.

Table 28: Equilibrium yield estimates for exploitable biomass (all legal-sized fish in blue zones in the area of the commercial fishery) and recruitment compensation ratio r = 10.

Results for exploitable biomass and ***r*** **= 10**		
Target biomass ratio	**Yield** (t)	**Harvest rate**, *U* (proportion per year)
0.40	1772	0.349
0.48	1600	0.272
0.58	1346	0.196
0.68	1059	0.135

Table 29: Equilibrium yield estimates for total adult biomass (all legal-sized fish in both blue and green zones in the Bioregions fished by the commercial fishery) and recruitment compensation ratio r = 10.

Results for total adult biomass and ***r*** **= 10**		
Target biomass ratio	**Yield** (t)	**Harvest rate**, *U* (proportion per year)
0.40	2007	0.644
0.48	1923	0.463
0.58	1694	0.310
0.68	1374	0.203

Table 30: Equilibrium yield estimates for exploitable biomass (all legal-sized fish in blue zones in the area of the commercial fishery) and recruitment compensation ratio r = 3.99.

Results for exploitable biomass and ***r*** **= 3.99**		
Target biomass ratio	**Yield** (t)	**Harvest rate**, *U* (proportion per year)
0.40	1411	0.386
0.48	1301	0.226
0.58	1171	0.165
0.68	894	0.115

Table 31: Equilibrium yield estimates for total adult biomass (all legal-sized fish in both blue and green zones in the Bioregions fished by the commercial fishery) and recruitment compensation ratio r = 3.99.

Results for total adult biomass and ***r*** **= 3.99**		
Target biomass ratio	**Yield** (t)	**Harvest rate**, *U* (proportion per year)
0.40	1490	0.431
0.48	1458	0.330
0.58	1319	0.234
0.68	1094	0.159

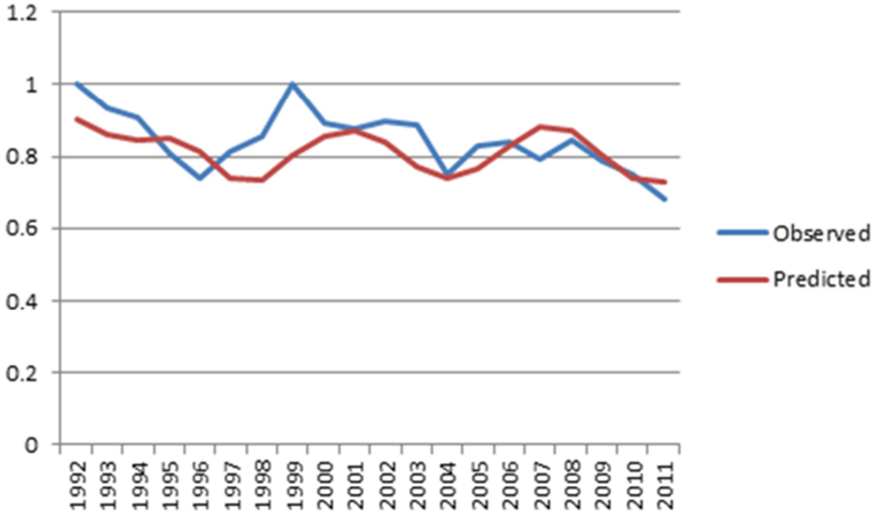


Figure 29: Coral trout observed (relative standardised catch rate) and predicted (stock model) commercial catch rates for the Cairns Subregion. The y-axis scale is relative to the observed 1992 value (= 1).

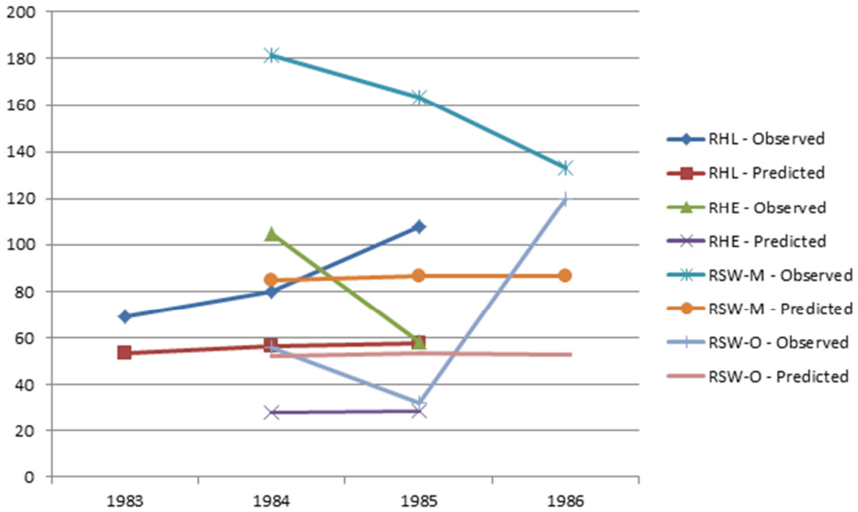


Figure 30: Coral trout observed (underwater visual survey estimate) and predicted (stock model) population density for blue Subbioregions RHL, RHE, RSW-M and RSW-O. The y-axis is scaled in numbers of legal-sized fish per hectare.

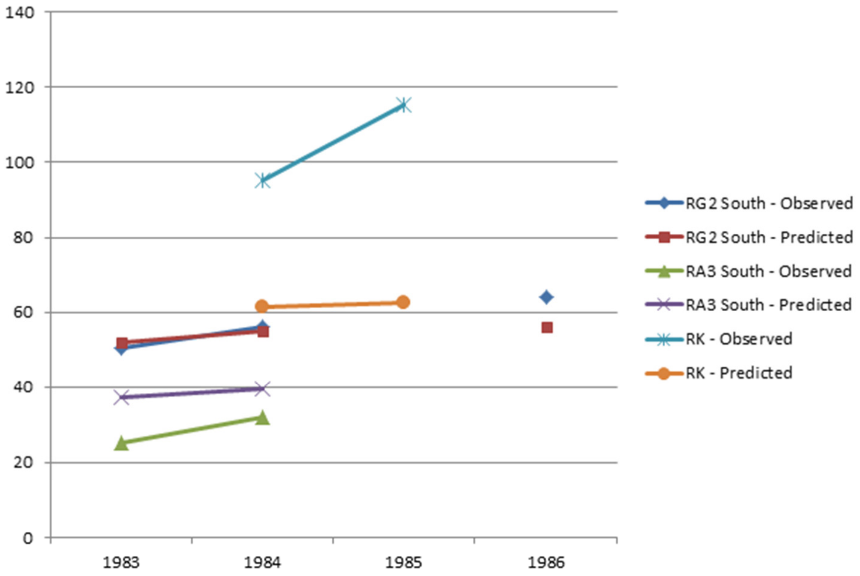


Figure 31: Coral trout observed (underwater visual survey estimate) and predicted (stock model) population density for blue Subbioregions RG2 South, RA3 South and RK. The y-axis is scaled in numbers of legal sized fish per hectare.

8. Discussion and recommendations

The stock assessment has highlighted the difficulties in assessing the coral trout population, and especially the poor correlation between catch rates and population size. The problems with catch rates are due in large part to the effect of tropical cyclones on coral trout catch rates, which are depressed for two to three years after a major cyclone but rebound after that time.

The connection between tropical cyclones and coral trout catch rates seems clear (Figure 22, page 71). However, the magnitude of the effect achieved by statistical modelling is relatively small. Analysis has shown statistically significant effects of cyclones (Figure 20, page 69), but has yet to make an impact on adjusting catch-rate time series by the amounts that are obviously required (Figure 23, page 72). Longer term, a research project funded by the Australian Government's Fisheries Research and Development Corporation and the Great Barrier Reef Marine Park Authority (FRDC project number 2013/020) is examining the effect of cyclones on the GBR, which could greatly improve the reliability of commercial catch rates.

A very valuable resource for the assessment has been the availability of underwater visual survey data to provide direct measurement of the numbers of coral trout present on the GBR. Without this data and the corresponding mapping data of the GBR to provide habitat areas, it would be impossible to assess this fishery with any degree of certainty.

The estimated population sizes from underwater visual survey data (Chapter 5) indicate that there is no current biological problem with the coral trout fishery. There is, however, a sizable economic problem, as indicated by relatively low catch rates in recent years and anecdotal information provided by commercial fishers. Fishing the GBR by hook and line (the only method compatible with the GBR's high conservation status) is costly, and profit levels for commercial fishers are reduced when catch rates are depressed.

This stock assessment has been conducted under the assumption that the capacity of the GBR to support fish populations (both predators such as coral trout, and the forage fish on which they feed) has not changed since serious data collection commenced in the mid-1980s. It has been pointed out to us (Rachel Pears, GBRMPA, personal communication) that live coral cover on the GBR has reduced by about 50% over that period (De'ath et al., 2012), which may have affected the GBR's carrying capacity. On the whole, available underwater visual survey data do not favour the hypothesis of a substantial reduction in carrying capacity for coral trout over the entire GBR (see, e.g., Figure 26, page 76).

The economic problem facing commercial fishers is compounded by the effect of social learning in coral trout (Chapter 2), whereby catch rates of even a lightly fished population are much lower than those of an unfished population. The fishers' response to maintain catch rates has been to spread themselves out over the suitable fishing locations in the GBR: the regional harvest sizes (Figure 18, page 62) are remarkably similar to the regional coral trout population estimates, with adjustments for the higher cost of travelling to the Far Northern Region (Cape York, Lockhart River, and Princess Charlotte Bay Subregions) and the Swains. In recent times fishers are still prepared to travel to these Regions far from port, and evidently believe that higher catch rates of coral trout will justify the extra expense.

Social learning by coral trout also exacerbates the problem of illegal fishing in green zones (marine protected areas) on the GBR. Indeed, the more effectively green zones are enforced, the more profitable it will become for fishers to risk fishing in them.

Because of the problems associated with catch rates, if catch rates are used to adjust the TACC, the adjustments should take into account long-term trends. While it is valid to adjust the total allowable commercial catch (TACC) for economic reasons as well as biological reasons, it should be borne in mind that the benefits of reducing a TACC would accrue when catch rates improve, such as after the fishery recovers from a major tropical cyclone. A lower total catch during these periods would make the fishery more economically robust by providing higher catch rates during the periods of depressed catch rates that will inevitably follow.

Current challenges facing the management of the coral trout fishery are a need to develop a new monitoring strategy and a need for quota management to be more responsive through adoption of quota setting rules. We offer the following points for consideration:

- Catch rates, both fishery-dependent (using data from fishers) and fishery-independent (using data from special scientific catch surveys) should be used with caution because they correlate poorly with population size due to the effects of tropical cyclones. Use of these data requires methods for effective catch-rate standardisation that are currently being researched and are not yet published.
- Adjustments to quota should take into consideration longer term trends in catch rates and, in contrast to other fisheries, are most important when catch rates are relatively high. The benefits of reducing the TACC include making the fishery more economically robust by providing higher catch rates during depressed periods such as after cyclones.
- Underwater visual surveys (UVS) are the most direct method of monitoring the abundance of coral trout, but are expensive and subject to large experimental errors stemming from low sample sizes (low total numbers of fish observed). The UVS data from the AIMS surveys provide an example of how much fish numbers from even quite large-scale UVS can vary from year to year (section 4.7 and Figure 26, page 76).
- Monitoring of age structures can detect trends in fishing mortality, but only in the long-term. It also is expensive as the aged fish could not be exported live.

We also make the following additional recommendation for fishery data:

- A central repository for UVS data on the GBR should be established as a matter of some urgency. These data were very expensive to collect at the time, and are still very valuable today. At present they are largely stored by individual scientists in various organisations. There is always the danger of failure of storage media or neglecting to keep up with technology; e.g., floppy discs often fail and computers that can read them have become rare. GBRMPA is the obvious candidate for holding the repository, as it paid for many of the surveys.

Finally, we raise the possibility that the Capricorn–Bunker Region could be analysed in detail in future, as a separate project to examine whether the fall in coral trout catch size since the early 2000s could have caused the rise in abundance that is apparent in the AIMS UVS data (see section 4.7, page 74). It may be the case, however, that the Capricorn–Bunker Region simply has different environmental drivers from the rest of the GBR.

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