

THE EFFECTS OF LINE FISHING ON THE GREAT BARRIER REEF AND EVALUATIONS OF ALTERNATIVE POTENTIAL MANAGEMENT STRATEGIES

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Non-Technical Summary

97/124	The Effects of Line Fishing on the Great Barrier Reef and Evaluations of Alternative Potential Management Strategies
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Objectives

- I. To understand the level of fishing that existing fish stocks and reef communities can sustain *via*:
 - Investigations of demographic characteristics of targeted species;
 - Experimental manipulations of fishing effort and management strategies;
 - Monitoring responses of non-target species to changes in fishing pressure, including responses of selected benthos and prey of target species; and
 - Relating responses of target and non-target species on experimental reefs to longer-term, broader scale information on abundances and (where appropriate) catch rates.
- II. To evaluate the efficacy of current management practices, specifically zoning strategies, with respect to the ecologically sustainable management of tropical reef line fishing.
- III. To document the limits of fishing induced changes in fish catch and other aspects of reef use that would be acceptable economically and socially to reef users.
- IV. To evaluate quantitatively potential management strategies for the future regulation of fishing such that fish stocks, ecosystem function, and yields to fisheries will be conserved.

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Executive Summary

The effects of reef line fishing on the productivity of targeted species and its impacts on other reef species on the Great Barrier Reef (GBR) have been poorly understood. Understanding the distribution, intensity, and effects of reef line fishing is essential for successful management of both fishing and other recreational and commercial activities in the GBR region, as well as for conservation of the GBR ecosystem.

The GBR Reef Line Fishery (RLF) comprises socially and economically important commercial, charter, and recreational fishing sectors. The fishery has been undergoing some change over the last decade, particularly manifest as considerable increases in effort and catch in the commercial fishery since 1995. These changes probably arise from several events, including changing management arrangements in other fisheries, the introduction of Dugong Protection Areas in in-shore areas, the process of reviewing management arrangements for the Reef Line Fishery and the development of lucrative export markets for live reef fish for consumption. Collectively, these influences have resulted in nearly 50% increase in commercial effort and 40% increase in catch since 1996. There also is potential for increased recreational fishing pressure along the GBR coast simply because of population growth and increased tourism. Management arrangements for the Coral Reef Fin Fish Fishery are now under review, with new management arrangements likely to regulate commercial effort in the fishery explicitly.

Conservation management of the GBR Marine Park also is undergoing significant change. The current zoning system is being substantially upgraded with the development of a comprehensive, adequate and representative system of no-take areas for biodiversity conservation of the GBR ecosystem – the Representative Areas Program. This revision is likely to increase the area of the GBR closed to reef line fishing. Realising the minimum regime of 20% of all GBR bioregions being ‘no-take’ will inevitably result in significant increases in the amount of coral reef habitat closed to the Reef Line Fishery in some areas. These factors, combined with limited historical information about the fishery or its main target species, present significant problems for planning appropriate management strategies of the fishery and the GBR World Heritage Area.

These factors, combined with limited historical information about the fishery or its main target species, present significant problems for the development of appropriate management strategies for the fishery and the GBR World Heritage Area. In this research, we have quantified some of the primary impacts of the RLF on targeted stocks and assessed secondary impacts on other components of the GBR ecosystem. We have assessed experimentally the degree to which area closure strategies are likely to have ameliorated those impacts. Finally, we evaluated the prospects for alternative mixes of strategies for conservation and fishery management in the region to realise the objectives of diverse stakeholders.

Surveys of areas that had been open and closed to fishing for over a decade showed that the two main target species of the RLF, the common coral trout and the red throat emperor, were significantly more abundant, larger and older in areas zoned Marine National Park ‘B’ (and so closed to fishing) than in adjacent General Use areas that have always been open to fishing. The magnitude of these differences varied regionally, from near-zero around Lizard Island to several-fold for some population characteristics in the southern regions of the GBR. The pattern in apparent ‘effectiveness’ of past closures matched closely patterns in the amount of fishing effort and catch and underlying patterns in the abundances of several harvest and non-harvest species. We present circumstantial arguments that this regional variation in the apparent ‘effectiveness’ of Marine Protected Areas is likely to reflect long-standing regional variations in the amounts of fishing and its impacts outside closed areas, rather than wholesale subversion of zoning strategies by high levels of poaching. That is, the lack of contrast between open and closed areas in the Lizard Region probably arises because the open areas are lightly fished, whereas the strong contrasts in the other regions arises because of relatively heavy fishing in the open areas in those regions.

Experimental manipulations of reef zoning status and fishing effort verified that fishing on reefs that had been closed historically reduced the abundances of target species on those reefs to levels similar to surrounding open reefs. In the absence of prior data with which to compare open and closed reefs before zoning was implemented, these manipulations provide the most convincing evidence that the Marine Park zoning strategies have been effective in protecting sub-populations of the fishery resource from the impacts of harvest. The protection of such refuges, with sufficient compliance, thus has the potential to sustain high biomass of reproductively mature populations of harvested species in spite of an active fishery on the GBR.

Indirect effects of line fishing on non-harvest fish were less conspicuous. Whilst differences existed between open and closed reefs in abundances of the prey of targeted species, the nature of the patterns varied regionally, through time and with species or species group. In some situations the patterns in abundance suggested that removal of a key predator (coral trout) by fishing might have allowed populations of some prey to grow on fished reefs, but the evidence was neither uniform nor convincing.

We have evaluated prospectively the relative merits for managers and stakeholders of alternative strategies for effort management and area closure on the GBR. We based these evaluations on a set of simulation models ('ELFSim') for the population dynamics and harvest of common coral trout on the GBR. The population dynamics model is spatially structured, depicting nearly 4000 reef-associated populations of coral trout inter-connected via larval dispersal. The reef-associated, post-settlement populations are age, size and sex structured and we allow for variation in most of the key demographic parameters, such as natural mortality, growth, recruitment, etc. The harvest model predicts the allocation of fishing effort over the GBR by three fishing fleets, parameterised with historical catch and effort data to represent the commercial, charter and recreational sectors of the RLF.

Objectives for the future status of coral trout populations and for the RLF were developed by a diverse set of stakeholders in the fishery and the GBR World Heritage Area, in association with the Reef Line Fishery Management Advisory Committee (ReefMAC). Contributing stakeholders included state and federal managers, commercial, charter and recreational fishers, conservation organisations, and researchers. Stakeholder objectives included preserving near-virgin biomass of coral trout on reefs closed to fishing, ensuring satisfactory levels of populations available for harvest, maintaining economically viable commercial catch rates and recreationally rewarding recreational catches of coral trout, and minimising variation in harvests from year to year. Quantitative articulations of these and other objectives were derived and agreed with stakeholders, together with associated performance indicators.

The same set of stakeholders advised on the mix of potential strategies to be considered for achieving their respective objectives. We were asked to compare the efficacy of three levels of fishing effort, ranging from half of 1996 levels to 1½ times 1996 levels, and three levels of area closure, ranging from current closures to nearly three times current closures. The outputs from these Management Strategy Evaluations provide comparative assessments of the likelihood that each of the stakeholder objectives will be met by each combination of effort control and area closure strategy. The results are not intended to prescribe which strategy mix should be adopted, but to provide a basis for stakeholders to negotiate such an outcome based on the degree to which different combinations of strategies meet their needs.

Harvest-related objectives (e.g., maintaining CPUE, increased chance of catching a large fish, preserving biomass available for harvest) were most likely to be achieved when effort was lowest under any area closure strategy, but were less likely to be achieved as increasing amounts of area were closed to fishing. The principle stock-conservation objective, represented by preserving the spawning biomass of the whole population, was most likely to be achieved by increasing the amount of area closure and was only relatively slightly impacted by increasing fishing effort within each area closure strategy.

Importantly, the observed increase in fishing effort in recent years is most likely to impact most negatively on the performance indicators for areas open to fishing, especially those reflecting what fishers would consider satisfactory performance of the fishery (e.g., catch rates and sizes of fish). The increase in area closures under the Representative Areas Program is likely to exacerbate the depreciation of fishery performance, but our results suggest that growth in fishing effort will be considerably more influential than changes in areas available to the fishery. Our results suggest that the currently elevated levels of effort (~1.5 time 1996 levels) will reduce significantly the prospects of fishers in all sectors realising their objectives in future years, irrespective of the inevitable increases in protected areas under the Representative Areas Program.

Reducing effort, conversely, is the strategy of those considered in our evaluations most likely to realise direct fisheries-related objectives. The conundrum in these results, however, is that the improved prospects from effort reduction would apply only to those fishers remaining in the fishery. We are unable to assess the magnitude of financial costs likely to be incurred by those fishers excluded through the effort reductions that would now be necessary to achieve the two lower effort scenarios we considered.

Changing effort had relatively little impact on most performance indicators for closed areas, especially conservation of spawning biomass of coral trout within Marine Protected Areas, even allowing for low levels of infringement of closed areas. The most effective mechanism by which to increase total spawning stock biomass over the GBR domain, therefore, was increasing the area closed to fishing, presuming that compliance with those closures was relatively high.

It is important to note that the status of coral trout populations in areas open to fishing remained relatively robust under all strategies we considered. For example, even under the most 'adverse' scenario of maximum effort constrained to the smallest fishable area, spawning biomass (in the open areas) remained above 50% of virgin spawning biomass and biomass available for harvest (*i.e.*, above the minimum legal size limit) remained above 30% of virgin available biomass. These statistics generally would be considered acceptable for a harvested stock. In large part, this is likely to be the consequence of the biologically precautionary minimum legal size limit on harvest of common coral trout, which ensures that most fish can spawn in at least one year before reaching harvestable size.

Sensitivity analyses for the simulations showed that the qualitative relationships among scenarios were robust to changes in model parameters. Accordingly, the conclusions about the *relative* merits of increasing or decreasing fishing effort or area closures are robust to most changes in model assumptions. It should be noted, however, that our evaluations relate only to the populations and harvest of common coral trout (*P. leopardus*). Though several other species harvested in the Reef Line Fishery are taxonomically close to *P. leopardus*, they are generally considered to be less abundant and longer lived than *P. leopardus* and their populations dynamics are perhaps less resilient to harvest than that of *P. leopardus*. Accordingly, conservative regulations for the harvest of these other species would be prudent at this stage.

This research has laid bare some of the inevitable trade-offs among different scenarios for managing the harvest of common coral trout by the RLF in the GBR World Heritage Area. Most importantly, the trade-offs have been assessed in relation to objectives and performance indicators specified by diverse stakeholders in the fishery and the World Heritage Area. We present the tradeoffs in ways that allow direct comparisons among disparate objectives, essentially providing a common currency for comparing performance across fundamentally different types of objectives. In so doing, we hope that the costs and benefits of different management options are more transparent to all stakeholders than might otherwise have been the case. We hope that such transparency aids in the negotiation of acceptable and effective future management arrangements for the Great Barrier Reef World Heritage Area and the Reef Line Fishery.

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Need

The potential for reef line fishing to significantly affect targeted species or indirectly impact other reef species is poorly understood. Understanding the distribution, intensity, and effects of reef line fishing is essential for successful management of both fishing and other recreational and commercial activities in the Great Barrier Reef (GBR) region, as well as for conservation of the GBR ecosystem.

The GBR Reef Line Fishery (RLF) has undergone substantial change since 1993, particularly manifest as increases in effort and catch in the commercial fishery since 1995. These changes probably resulted from several events, including restrictions introduced for other fisheries (spanner crab, in-shore net and trawl) by the Queensland Fisheries Management Authority (QFMA) and Queensland Fisheries Service (QFS) on behalf of the Queensland Government, the introduction of Dugong Protection Areas in in-shore areas gazetted by the Great Barrier Reef Marine Park Authority (GBRMPA) on behalf of the Commonwealth Government, the announcement of a major review of management arrangements for the RLF and the development of lucrative export markets for live reef fish for consumption. Indirect indicators of the attractiveness of the RLF to commercial fishers include the 2-3 fold increase in the market value of reef line fishing licences since 1994, the acquisition of multiple vessels by some operators, purchases of large, purpose built or re-fitted vessels for the live fish sector of the fishery and increased participation in the commercial fishery by licence holders who previously had not harvested reef fish. There is also considerable potential for increased recreational fishing pressure simply because of population growth and increased tourism. These factors, combined with the dearth of historical information about the fishery or its main target species, present significant problems for formulating appropriate management strategies for the fishery and the GBR World Heritage Area.

Conservation management of the GBR Marine Park also is undergoing significant change with the development of a comprehensive, adequate and representative system of no-take areas likely to increase the area of the GBR closed to reef line fishing. Management arrangements for the Coral Reef Fin Fish Fishery are under review, with new management arrangements likely to regulate commercial effort in the fishery explicitly. There is an immediate need for quantitative assessments of the impacts of line fishing on targeted stocks and other fishes and for prospective evaluation of the value of proposed management strategies in relation to economic, social, biological and conservation objectives for the GBR and its use. In "Research Needs and Priorities" (1996), key topics for research related to the Reef Line Fishery flagged by the QFMA included: Appraise management measures for the sustainable use of reef fish; Determine an effective mix of measures for reef fish management planning, including fishery dependent and independent monitoring; Determination of the size of stocks of common coral trout; Determination of the proportion of blue-spot trout in the reef line catch; and assess regional catch rates of red-throat sweetlip.

The CRC Reef Effects of Line Fishing (ELF) Project was designed to produce results of direct relevance to these and other needs, and in so doing contribute management-relevant information for the GBR RLF. Two components of the ELF Project, the ELF Experiment and Management Strategy Evaluations (MSE) for the RLF, were specifically designed to meet several of the above priority needs. This report documents key results to date from these two components of the ELF Project.

Objectives

Below are the objectives for the CRC Reef Effects of Line Fishing Project. Specific subsidiary objectives of particular sub-tasks are listed in Box 1.

- I. To document the distribution and intensity of reef-based fishing catch and effort and patterns in relative abundance of fish stocks.
- II. To understand the level of fishing that existing fish stocks and reef communities can sustain *via*:
 - a. Investigations of demographic characteristics of targeted species;
 - b. Experimental manipulations of fishing effort and management strategies;
 - c. Monitoring responses of non-target species to changes in fishing pressure, including responses of selected benthos and prey of target species; and
 - d. Relating responses of target and non-target species on experimental reefs to longer-term, broader scale information on abundances and (where appropriate) catch rates.
- III. To evaluate the efficacy of current management practices, specifically zoning strategies, with respect to the ecologically sustainable management of tropical reef line fishing
- IV. To document the limits of fishing induced changes in fish catch and other aspects of reef use that would be acceptable economically and socially to reef users.
- V. To evaluate quantitatively potential management strategies for the future regulation of fishing such that fish stocks, ecosystem function, and yields to fisheries will be conserved.

This report addresses objectives II – V, or substantive components of them.

Achievement of Objectives

The distribution and intensity of fishing has been described from analyses of compulsory logbook data from the commercial and charter sectors of the fishery provided by the Queensland Fisheries Management Authority and Queensland Fisheries Service (QFS). Interpretation of the analyses has been supplemented from interviews, voluntary research logbook programs and observer programs as part of the ELF Project. Characteristics of the small-boat recreational fishery have been provided to the project by QFS.

Objectives II and III have been achieved via successful implementation of the ELF Experiment, involving changes in reef zoning status (opened or closed to fishing). Responses to the manipulations have been assessed by Underwater Visual Surveys and line fishing Catch Surveys, the latter also facilitating collection of biological data for harvest species. Several post-graduate research projects have aided in achieving these objectives.

Objective IV has been achieved via a series of formal and informal workshops with diverse stakeholders in the GBR and the RLF. Contributing stakeholders included Marine Park and Fishery managers, commercial, recreational and charter fishers, fish marketers, conservation lobbyists and researchers. Achieving this objective was central to completion of objective V.

Objective V has been achieved in part through the achievement of the other objectives, but substantially via the development of sophisticated computer simulations of the population dynamics and harvest of coral trout over the extent of the GBR. The Effects of Line Fishing Simulator (ELFSim) captures the spatial and biological complexity of the GBR and the diversity of harvest behaviours employed by commercial, recreational and charter fishers of the resource. ELFSim has been used to assess the prospects of achieving diverse stakeholder-specified objectives through an array of combinations of proposed fishery and conservation management strategies.

Box 1: Sub-task Specific Objectives

1. Catch Surveys

- a) To establish structured catch surveys in which the fishing characteristics, personnel, vessel, and data collection are standardised across all reefs and times from which to estimated a) differences among reefs historically closed or open to fishing; b) regional patterns in catch rates; c) effects of simulated increases in fishing pressure and protection from fishing in the ELF Experiment;
- b) To compare the relationship between CPUE data from line and spear fishing and underwater visual survey (UVS) and the capacity of each method to detect a known decrease in fish abundance;
- c) To collect size, otolith, gonad and gut samples from fish caught during the ELF Experiment for studies of population dynamics of major target species.

2. Visual Surveys

- a) To provide fishery independent underwater visual estimates of population densities of target fish species, prey species and other reef organisms on the reefs involved in the ELF Experiment;
- b) To monitor recruitment of the main target species for the RLF to the experimental reefs.

3. Demographic Studies

- a) To derive age-structure data as a primary indicator of the effects of fishing on reef fish stocks;
- b) To use age and length structure data to estimate growth and mortality rates and provide measures of recruitment rates to reef fish stocks;
- c) To provide catch-at-age data from which to model population dynamics of target species.

4. Fleet Dynamics and Fishery Information

- a) To establish and maintain cooperative links with the recreational, commercial and charter boat line fishing community to: a) increase fishing effort on the manipulation reefs in the ELF Experiment; and b) obtain detailed fleet catch and effort information from the reefs involved in the ELF Experiment;
- b) To identify and compare factors which affect the fleet dynamics and on site fishing practices in the RLF within and among regions of the GBR.

5. Assessment of Bias in Line-caught Samples

- a) To compare age and size structures of populations of common coral trout (*Plectropomus leopardus*), sampled by spear fishing, among reefs open and closed to fishing and among regions of the Great Barrier Reef (GBR);
- b) To examine the size and age selectivity of line fishing by comparing age and size structures of samples of common coral trout taken from reefs open and closed to fishing by spear and line fishing;
- c) To examine the relative biases of line and spear fishing as techniques for sampling reef fish.

6 Modelling and Management Strategy Evaluation

- a) To develop and parameterise population dynamics models of the primary target (fished) species (*P. leopardus*) that are appropriate for the spatially structured GBR and that are amenable to application to secondary target species;
- b) To develop models of effort allocation, and where possible, fleet dynamics of different components of the fishery, including consideration of the commercial, recreational and charter fishing sectors, that are appropriate for the dynamics of the reef line fishery in the spatially complex GBR;
- c) To provide an integrated framework, based on the population and effort dynamic models, for evaluating alternative management scenarios to quantify the likely trade-offs between conflicting objectives and their consequences for the coral trout fishery and the fish stocks;
- d) To develop with stakeholders in the GBR line fishery operational objectives and potential future management strategies relevant and amenable to evaluation from biological and fishery information;
- e) To apply the management strategy evaluation framework to evaluate risks and benefits of alternative strategies for managing the RLF on the GBR against stakeholder objectives for the fishery.

7. Coordination and liaison

- a) To coordinate the implementation of all components of the ELF Project in a coherent manner across multiple institutions and with stakeholder groups;
- b) To manage peer review of the ELF Experiment;
- c) To coordinate liaison with researchers and stakeholders across a range of levels, from senior policy fora to the general public;
- d) To manage the coherent collation and archival of data and biological material from the ELF Project

General Introduction

The Great Barrier Reef (GBR) lies off the Queensland coast (Australia) and comprises the world's largest coral reef archipelago, spanning over 15° of latitude from just south of Papua New Guinea (~9°S) to just north of Fraser Island (~25°S) (Fig. 1). Most of the GBR was declared a multiple use marine park in 1975 (*Great Barrier Reef Marine Park Act 1975*) and inscribed on the World Heritage Area list in 1981 (Australian Heritage Commission Act 1975, World Heritage Properties Conservation Act 1983). Thus, the GBR has been formally recognised as having special significance nationally and internationally, predominantly for its natural features, and in need of conservation “in perpetuity” (Kenchington 1990). In the defining legislation, however, the GBR Marine Park was gazetted for multiple use, excluding only mining. Existing uses were recognised also when the GBR was given World Heritage listing. The facilitation of such multiple use, in so far as it is consistent with conservation of the natural features of the GBR, is integral to the management of the GBR Marine Park (GBRMP) and World Heritage Area (GBRWA). Thus, the region supports a thriving tourism industry as well as port and shipping activities and a diversity of wild harvest fisheries (Robertson 1997).

Figure 1: Location map for the Great Barrier Reef indicating general structural features such as geographic extent, varying proximity to coastal influences and population centres, and fragmented structure. The boundary of the GBR Marine Park is also shown, with the boundaries between management sections indicated (lines running from coast to seaward boundary of the park).



In the absence of mining activities, fishing is the principal extractive use of the GBR and is of particular concern because of the potential for fishing to impact negatively on the heritage and ecological values for which the GBR has been recognised. Management of fishing activities, therefore, emphasises consistency with the conservation requirements of the *Great Barrier Reef Marine Park Act* (1975), World Heritage Values, and principles of ecological sustainability, as well as traditional expectations of securing yields, recreational enjoyment and community benefit (Driml 1994, 1999, Hundloe 1985, Kenchington 1990).

Fishing has occurred in the Great Barrier Reef (GBR) Region for decades, from well before the declaration of the GBRMP and GBRWA. It is now a multi-million dollar industry deriving regional income from commercial fishing (mainly trawling for prawns, line fishing and netting for fin-fish, crabbing, and collection fisheries for aquarium fish, Beche de Mer, *Trochus*, and crayfish), local recreational fishing, and tourism (including game-fishing). Line and spear fishing are the activities that have the greatest potential to affect the biological communities on coral reefs *per se* of the GBR, as opposed to inter-reef habitats.

The Demersal Reef Line Fishery

The RLF is comprised of three main sectors: a commercial sector, a charter fishing sector, and a private recreational sector. Fishers in all sectors use basically similar gears, typically consisting of single baited hooks on heavy line on rod and reel or hand reel. The RLF is multi-species in all sectors, with over 125 species or species groups being reported by the commercial sector (Mapstone *et al.*, 1996a,b,c) and at least 108 species or species groups reported in the catch from the charter and recreational sectors (Green *et al.* in prep, Higgs 1999, Morgan 1999). The primary targeted species in the fishery are the common coral trout (*Plectropomus leopardus*) and the red throat emperor (*Lethrinus miniatus*) (Mapstone *et al.* 1996a,b,c, Queensland Fisheries Management Authority (QFMA) 1996, Queensland Fisheries Service (QFS) 2002). The generic group 'coral trout', comprising three main harvested species and four lesser species, are estimated to comprise approximately 35-55% of the catch by commercial fishers, approximately 20-25% of the charter catch and 15-20% of the recreational catch of demersal reef fish from the GBR region. *L. miniatus* comprises about 20% of the commercial catch, up to 40% of the charter fishing catch and about 30-35% of the recreational demersal catch annually. Other emperors, especially spangled emperor (*L. nebulosus*) and snappers (mostly red emperor, *Lutjanus sebae* and large and small mouth nannygai, *L. malabaricus*, *L. erythropterus*) and serranids other than coral trout are the next most targeted demersal species, with their relative importance varying among sectors and among regions within sectors (Green *et al.* in prep, Higgs 1999, Mapstone *et al.* 1996a, 1997, QFS 2002).

The Commercial Sector of the Reef Line Fishery

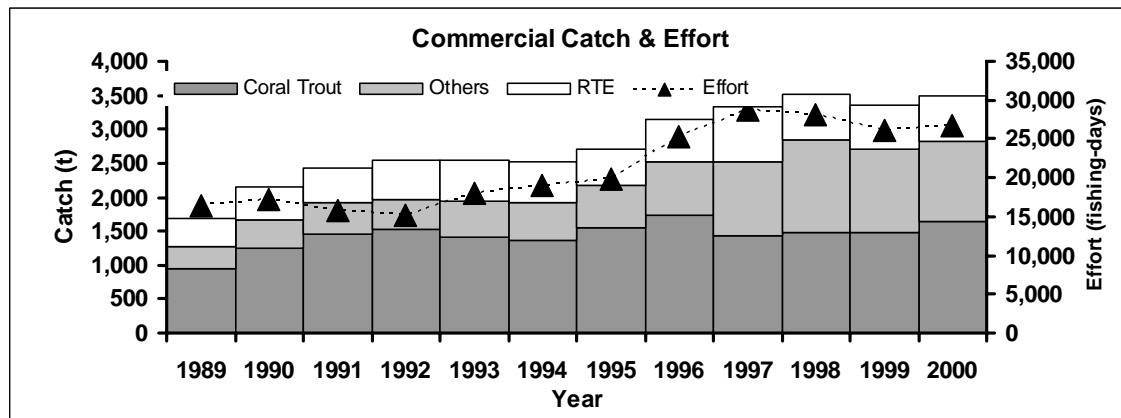
The bulk of the commercial fishery occurs from 4-7m dories based on 8-19m primary vessels, though fishing also occurs from the primary vessels. The most active commercial reef-line operations have 2-5 dories per primary vessel, whilst the majority of operations have no dories and report relatively little catch and effort (Mapstone *et al.* 1996a).

The commercial Reef Line Fishery has been undergoing some change since 1993, particularly manifest as considerable increase in effort since 1995. These changes probably resulted from several events, including restrictions on other fisheries (spanner crab, in-shore net, and trawl), the introduction of Dugong Protection Areas in in-shore areas, the announcement of a major review of management arrangements for the RLF, and the development of lucrative export markets for live reef fish for consumption. Each of these events has the potential to have prompted activation of latent effort or the transfer of effort from other fisheries to the RLF and corresponding increases in commercial fishing effort.

The value added to the commercial catch as a result of high market values for live fish (beach prices up to \$45/kg in 1996 and \$58/kg in 2000), in particular, provides increased financial incentives to participate in the RLF. Indirect indicators of the attractiveness of the RLF to commercial fishers include the 2-3 fold increase in the market value of reef line commercial fishing licences since 1994, the acquisition of multiple vessels by some operators and purchases of large, purpose built or re-fitted vessels for the live fish fishery.

Compulsory logbooks of catch and effort were introduced for commercial fishers only in 1988. Between 1988 and 1994 there were between 361 and 416 commercial line fishing operations reporting catch of demersal coral reef species from the GBR. The 20% most active operations accounted for over 60% of the catch in most years and the majority of operations reported less than 1 tonne of catch annually (Gwynne 1990, Mapstone *et al.*, 1996a,b,c, Trainor 1991). Total annual harvest of all species (including pelagic species) by the commercial RLF over that period was between 2,873 and 3,725 tonnes, arising from 16,000 – 18,500 fishing days (Mapstone *et al.* 1996a,b). Since 1994, however, reported effort and catch have increased substantially (QFS 2002, Williams 2002; Fig. 2), reaching their highest levels of nearly 40,000 fishing days by over 700 operations taking 4,400 tonnes of demersal reef fish from all Queensland waters in 2001 (QFS 2002).

Figure 2: Annual commercial catches of **coral trout** (all species of *Plectropomus* spp. and *Variola* spp.), red throat emperor (*Lethrinus miniatus*, RTE) and other demersal reef fish (**others**) from the GBR Marine Park between 1989 and 2000 and the number of reported fishing days from which the catch came.



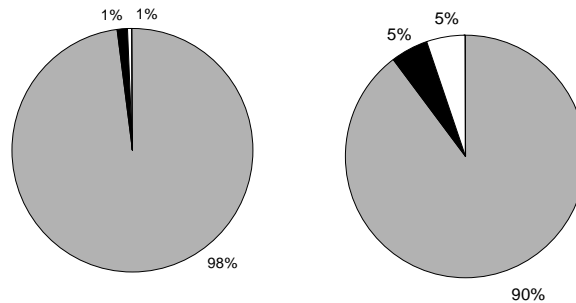
The Charter and Recreational Sectors of the Reef Line Fishery

Recreational fishing occurs from both privately owned boats, mostly outboard powered, or from charter vessels that take single or multi-day trips to the GBR. Prior to 1996, information about the private recreational fishing fleet came only from boat registrations and mostly recall data obtained from fishers during various creel, telephone, or postal surveys (Blamey and Hundloe 1993, Craik and Fallows 1980, Higgs 1996). The longest history of data derives from line and spear fishing club competition records (Higgs 1996, Nakaya 1998), but recent population surveys indicated that those data represent at most 6% of the recreational fishing population (Morgan 1999).

Blamey and Hundloe (1993) estimated that the total recreational catch in the GBR region in 1989-90 was between 3,500 and 4,300 tonnes from approximately 210,000-270,000 'fishing trips to sea' by 160,000 recreational fishers in private vessels. Only 5.4%-13.5% of these trips (depending on region), however, went 'off-shore', and smaller proportions visited the 'GBR proper'. More recent research and surveys sponsored by the Queensland Fisheries Management Authority (QFMA) and Queensland Fisheries Service (QFS) indicated similar distributions of recreational fishing between near shore and off-shore waters (Fig. 3) and indicate that recreational anglers took approximately 2,000t of demersal reef fish per year in the late 1990s (Higgs 1996, 1999, Mapstone *et al.* 1997, QFS 2002, Williams 2002). There is considerable potential for increased recreational fishing pressure along the GBR coast, because of population growth and increased tourism. Financial constraints such as the cost of fuel for long trips to the off-shore reefs, however, probably dampen effects of population growth on off-shore reefs.

Fishing on charter trips occurs mostly from the main charter vessel but dories are also used on some extended charters. There were 415 operators of charter vessels in the GBR region holding permits to take clients reef fishing in 1995, though most of these probably did 'fishing trips' infrequently. Green *et al.* (in prep) estimated that only about 120 charter fishing operations regularly took clients to sea predominantly for fishing in 1999-2000. Fishers on charter trips to the GBR fished for an estimated 11,030-22,880 line-days between 1996 and 1998 and kept 50-51 tonnes of coral trout and related species of demersal reef fish (QFS unpublished data). Compulsory logbooks for charter vessel operators were introduced only in 1996-97.

Figure 3: Percentages of recreational boating trips from the Ingham area in May-July 1995 that went to near shore islands (black), off-shore reefs of the GBR (white), or in-shore areas (grey). The pie on the left was derived from interviews at boat ramps, that on the right from voluntary angler logbooks (Mapstone *et al.* 1997).



Distribution of Reef Line Fishing

Line fishing occurs over the entirety of the GBR, though by far the majority of catch and effort is in the southern half of the GBR, south of Cairns (Blamey and Hundloe 1993, Gwynne 1990, Higgs 1996, Mapstone *et al.* 1996a,c, Trainor 1991). The greatest small-boat recreational effort on the off-shore GBR occurs between north of Townsville and just north of Cairns (Blamey and Hundloe 1993, Higgs 1996, 1999), the majority of charter vessel effort is centred on Gladstone, Mackay, Airlie Beach, and Cairns (Green *et al.* in prep) and the bulk of the commercial effort occurs south of Cairns and north of Rockhampton (Mapstone *et al.* 1996a) (Fig. 4).

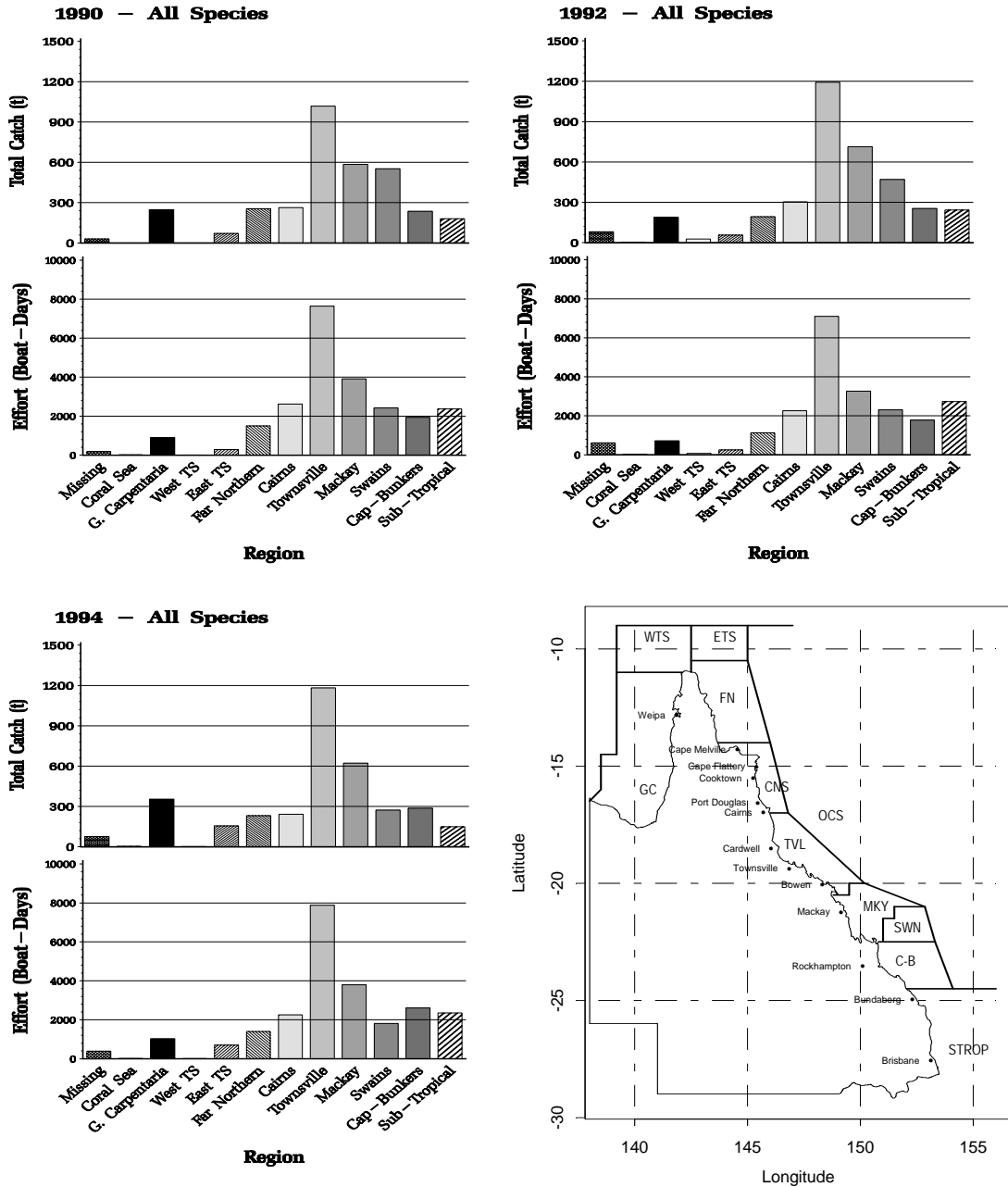
Management Arrangements Affecting the Reef Line Fishery

The management régime for the GBR region is complex, with responsibility for conservation management and fisheries management being divided constitutionally, legislatively, and operationally between the Australian Commonwealth and Queensland Governments. Management responsibilities for the GBR Marine Park and World Heritage Area are vested by the Commonwealth with the Great Barrier Reef Marine Park Authority (GBRMPA), a Commonwealth Statutory Authority (GBR Marine Park Act 1975). The GBRMPA has responsibility for management for 'protection and wise use in perpetuity', but explicitly does not have authority for fisheries management. Consistent with the Off-shore Constitutional Settlement (1981), responsibility for managing fisheries is delegated to the Queensland Government, which administers that mandate through the Queensland Fisheries Service⁸ (QFS) (Queensland Fisheries Act 1994, 1999).

The GBRMPA has to date employed a strategy of zoning for conservation management of the GBR. Under zoning plans developed by the Authority and enacted by the Federal Parliament, different areas of the marine park are designated for different types of permissible use, ranging from most extractive activities (General Use zones) to 'look but don't touch' (Marine National Park-B zones) and 'no-go' areas (Preservation zones). Although not specifically designed as fisheries management strategies, these zoning strategies effectively regulate fishing through the implementation of area closures that protect various species from harvest and impose broad restrictions on fishers' access to stocks. Although the 2000km length of the GBR Marine Park has been divided into four major sections for zoning (Fig. 1), to date the zoning plans applied to each have been very similar and have not resulted in region-specific management.

⁸ Management of fisheries in Queensland was through the Queensland Fisheries Management Authority (QFMA), a statutory Authority, between 1993 and 2000 but from July 1, 2000, the QFMA was dissolved and responsibility for managing fishing in Queensland was transferred to the Queensland Fisheries Service, within the portfolio responsibilities of the Department of Primary Industries.

Figure 4: Annual total effort and total catch of all line-caught species as reported in the compulsory commercial logbooks in each region in 1990, 1992 and 1994. Region boundaries are shown in the figure at bottom right, with the key to region abbreviations in the table below the map. (Adapted from Mapstone *et al.* 1996a).



Key to Regions Indicated on Map			
GC	Gulf of Carpentaria	MKY	Mackay
WTS	Western Torres Straits	SWN	Swains
ETS	Eastern Torres Straits	C-B	Capricorn Bunkers
FN	Far Northern	OCS	Off-shore Coral Sea
CNS	Cairns	STROP	Sub-tropical
TVL	Townsville		

The RLF also is managed uniformly over the length of entire GBR (QFMA 1999, QFS 2002), although since April 1999 reefs in the Torres Strait, north of 10°20' S, fall under a different jurisdiction from the remainder of the GBR (AFMA 2001). The prospect arises, therefore, that management arrangements might become non-uniform if changes are legislated in either jurisdiction without reflection in the other. The QFS, and formerly the QFMA, impose a number of explicit fisheries management strategies in Marine Park zones of the GBR where line fishing is allowed (QFMA 1999, QFS 2002). Minimum legal size limits for several of the main target species apply to all sectors of the fishery, as do gear restrictions.

Commercial fishing is managed via a two-tiered limited entry licensing scheme combined with vessel length constraints common to both. The two tiers were defined in 1993 in relation to the evidence of prior participation in the RLF and are implemented as limits on the numbers of dories that can be used in conjunction with each primary vessel. Licences with an 'L2' endorsement are allowed up to seven dories per primary vessel, with the number specific to each licence and fixed as of 1993. L2 endorsed primary vessels support an average of 3-4 dories. 'L3' endorsed licences allow fishing from at most one dory in addition to the primary vessel. There are no restrictions on the numbers of lines or fishers per operation, except those imposed by survey conditions of the primary vessel. In addition to the 'L2' or 'L3' endorsements, most commercial licence packages carry multiple other endorsements, including one or more of the available line, net and crab, spanner crab, and trawl endorsements. Movement among fisheries is at the discretion of the master fishermen, providing they hold the appropriate endorsements.

Recreational fishers are regulated by per-person 'in possession' total bag limits of 30 fish per angler for reef fish, which includes smaller bag limits for several species or species groups. Fishers on day trips are allowed 1 bag per person per day, but fishers on extended trips to sea (> 48 hours at sea) on charter vessels are allowed 2 bags each for the trip. Charter fishing operators are regulated by permits issued by the GBRMPA (for operation in the GBRMP) and QFS (for operation in Queensland waters), but otherwise there is no license or permitting system in place for recreational fishers. Management arrangements for all sectors of the RLF under Queensland jurisdiction (which excludes the Torres Strait) are currently under review.

The relative shortage of historical information about the fishery, its main target species or its environmental impacts (see Mapstone *et al.* 1997 for review) combined with changes in the fishery since the mid-1990s present major issues for planning appropriate management strategies for the fishery and the GBR WHA (QFS 2002).

The Effects of Line Fishing Experiment

Introduction

The Effects of Line Fishing (ELF) Experiment has its genesis in a research proposal to the Great Barrier Reef Marine Park Authority (GBRMPA) in 1988 (Mapstone *et al.* 1988) and a workshop convened jointly by the GBRMPA and the Queensland Department of Primary Industries (QDPI) in the February 1989 under the advice of the Advisory Committee on Research into the Effects of Fishing on the Great Barrier Reef (Craik *et al.* 1989). Trawl and line fisheries were considered to be those in greatest need of information for management in the GBR Marine Park and a major recommendation from the workshop was the development of a large scale manipulative experiment to examine the ecosystem effects of these fisheries and derive important parameters for their management. In 1990, the GBRMPA commissioned a study to examine the feasibility of such an approach and recommend experimental designs and necessary preliminary research (Walters and Sainsbury 1990). Although Walters and Sainsbury (1990) recommended a study to examine the joint and interactive effects of trawling and line fishing, subsequent research bifurcated into separate studies of the two fisheries. Whilst research into the effects of prawn trawling was championed by the CSIRO (then) Division of Fisheries, research into the effects of line fishing lacked an institutional focus until 1993. The establishment of the Cooperative Research Centre for the Ecologically Sustainable Development of the Great Barrier Reef¹ (CRC Reef) in 1993 provided a vehicle for the development of a comprehensive program of research into line fishing on the GBR, including, but not restricted to, the experimental approach discussed since 1989 (The Effects of Line Fishing (ELF) Project, Box 2).

Box 2: Background to the Effects of Line Fishing Project

The ELF Project commenced in 1993 as a core task of the CRC Reef to fill some of the gaps in information about the RLF and its impacts on the GBR. The ELF Project involves collaborations among CRC Reef funded researchers and researchers from the Australian Institute of Marine Science, CSIRO, James Cook University, Queensland Department of Primary Industries, Sea Research and the University of Queensland, managers from the GBRMPA and QFS and fishers from the commercial, charter and recreational sectors of the RLF. Cash funding for the ELF Project is predominantly from the CRC Reef, but significant funds are provided also by the Fisheries Research and Development Corporation (1997-2004), the GBRMPA (1997-2000), QFMA, and James Cook University. Substantial in-kind funding derives from the participating research institutions, management agencies, and sectors of the RLF.

The project involves six main research areas: i) Documenting the fishery history from catch and effort data, oral history, and structured interviews; ii) Monitoring the fishery from contemporary research and management logbooks, research surveys, student projects and collaboration with other agencies; iii) Documenting biological characteristics of the targeted stocks and other fish impacted by the fishery, either as by-product or by-catch; iv) Testing responses of target stocks, their prey, and fishing fleets to changes in fishing pressure through a large-scale controlled experiment; v) Evaluating the strengths and weaknesses of alternative potential management strategies for the GBR Marine Park and the RLF through dynamic computer models; and vi) Liaison and extension of research results to stakeholders and into policy fora. Extension activities include representation on Management Advisory Committees, presentations to peak bodies and management agencies, face-to-face discussions with stakeholders at ports, public meetings and trade fairs, publication of articles in the popular and technical press and the distribution of a quarterly newsletter to ~2000 recipients. The ELF Project currently has support until June 2006.

Several studies were done prior to 1994 as pilot studies in anticipation of the eventual implementation of a large scale experiment to examine the effects of line fishing on the GBR. These projects included methodological studies (Cappo and Brown 1996, Mapstone and

¹ The tenure of this CRC was 1993-2000. It has subsequently been revised and re-funded as the CRC for the Great Barrier Reef World Heritage Area, with tenure extended to 2006.

Ayling 1998), exploration of key assumptions underlying Walters' and Sainsbury's recommendations (Davies 1995 a,b), and biological studies of the key target species of the Reef Line Fishery (Ayling and Ayling 1992a,b, Brown *et al.* 1994, Ferreira and Russ 1994, 1995, Russ *et al.* 1995, 1996, Samoily and Squire 1994, Samoily 1997, Williams and Russ 1994), and surveys of the effects of changed zoning of reefs (Ayling and Ayling 1992b, 1994, 1995, Bienssen 1989, Brown *et al.* 1993, 1996, Mapstone *et al.* 1996d). In 1994 a second design evaluation was done in the light of the more recent information provided by these and other studies and the logistic and financial resources available under the CRC programme (Mapstone *et al.* 1996e, Campbell *et al.* 2001). The feasibility and desirability of a large-scale experiment to examine the effects of line fishing on the GBR were confirmed and a revised experimental design recommended (Mapstone *et al.* 1996e, Campbell *et al.* 2001).

During 1995 and 1996 the recommended experimental design was canvassed widely among interested stakeholders to assess its legal, political, and logistic feasibility. The consultation process involved a combination of public and stakeholder-specific meetings, explanatory articles in the popular press, discussions with a range of multi-sectoral regional management advisory committees, a formal, nationally advertised public review process, and consideration of the proposed experiment by the Federal Parliament, the Australian Heritage Commission, and the Commonwealth Environment Protection Group. Inevitably, some aspects of the suggested implementation of the experiment were changed because of logistic and political constraints highlighted during the consultation phase², but the basic recommendations for the ELF Experiment received wide support.

The ELF Experiment was opposed by some conservation groups on the grounds that it involved opening to fishing some reefs that had been closed to fishing previously (Marine National Park B zones, 'green' reefs) and opposed by some fishers because it involved the additional closures of reefs available to the fishery (General Use (GU), 'blue' reefs). The experiment was supported, however, by the relevant management agencies (GBRMPA and the QFMA), representative bodies of both recreational and commercial fishing sectors (respectively, SUNFISH and the Queensland Commercial Fishermen's Organisation, QCFO), the Queensland Conservation Council, the Australian Marine Conservation Society, the Australian Marine Science Association, researchers, Regional Marine Resource Advisory Committees, the Reef Line Fishery Management Advisory Committee (ReefMAC), and recreational and commercial fishers and charter vessel operators along the GBR Coast.

Implementing the ELF Experiment involved changes to the GBRMPA zoning provisions for some of the reefs involved, and those changes were outside the scope of the GBRMPA's mandate allowed by the existing Zoning Plans approved by the Federal Parliament. Accordingly, the experiment could proceed only if the Federal Parliament amended the relevant Zoning Plans to allow the unusual changes in current management provisions. The necessary amendments to the GBR Zoning Plans were passed by the Federal Parliament in November 1996 and the decision by the GBRMPA to implement the amendments was gazetted on January 8, 1997.

The primary objective of the ELF Experiment is to provide, through deliberate manipulations of fishing pressure and reef closures to fishing, field data from which to derive key underlying parameters for the evaluation of biological and fishery responses to alternative management scenarios. In particular, the contrast in population properties induced by the experiment should provide the empirical basis for estimating:

1. Reef-specific population size and estimates of sustainable harvest;
2. The coefficient of catchability for key target species; and
3. Fishing mortality and natural mortality of target species.

These parameters will be estimated from:

² In particular: a) the timing of the manipulations was condensed such that manipulations of fishing pressure and reef closure were to be applied over 2 years instead of 4; and b) suggestions of sponsored fishing by clubs and/or spear fishers and subsidised commercial fishing to force down populations on previously fished reefs were abandoned.

4. Empirical estimates of demographic responses of the main target species to changes in fishing mortality;
5. Comparisons of the behaviours of catch per unit of effort and underwater visual counts as indices of relative stock density, specifically in relation to changes in stock size under changed harvest régimes.

In addition, the experiment will provide direct evidence about:

6. Regional patterns in the effects of prior closure of reefs to fishing on catch rates, population densities, age, size and sex structures of target species of the RLF;
7. The dynamics of recovery of reef fish populations after protection from fishing; and
8. Changes in prey species abundances following reductions in predator density by fishing.

Because of the controversy about the experiment and its conduct in a World Heritage Area, the merits of continuing with the work after 1998 were reviewed by an international panel. This review followed the first set of experimental manipulations of zoning and fishing on selected reefs and preceded the proposed second set of such manipulations. The review panel recommended that the experiment should proceed to completion (Mapstone *et al.* 1998a, Davies *et al.* 1998).

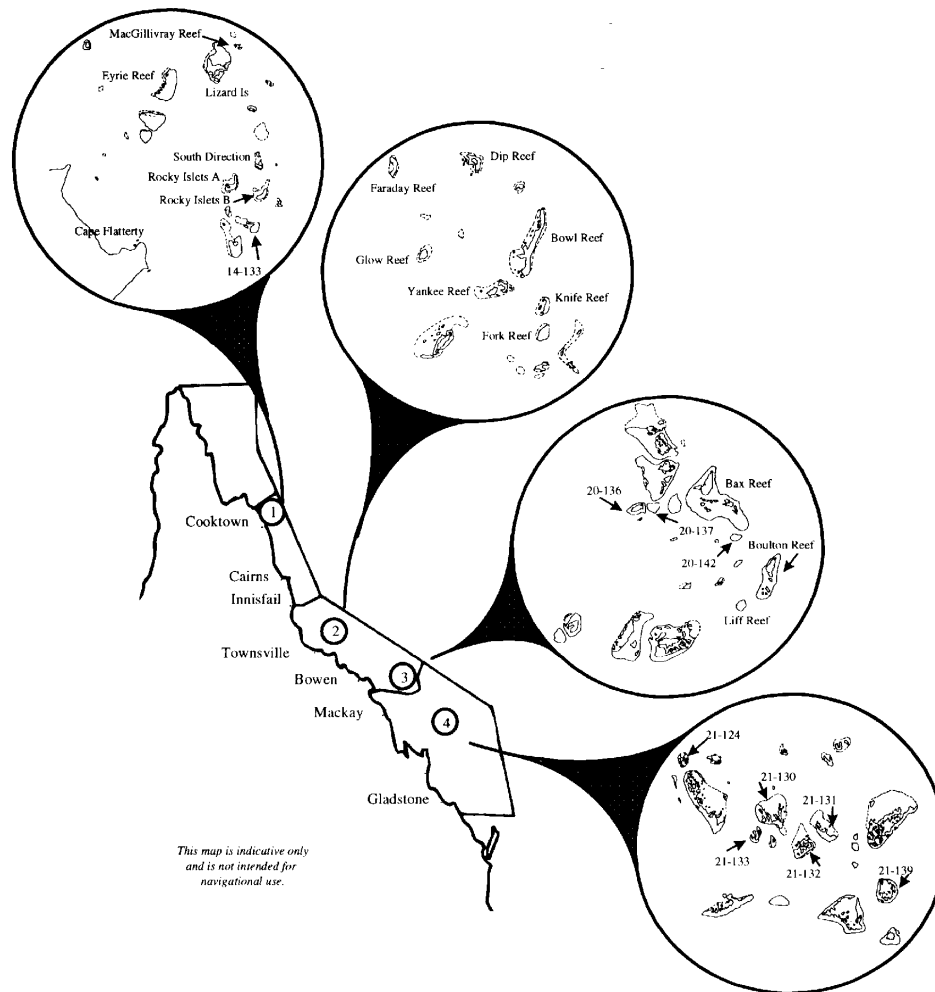
The ELF Experiment is not yet complete, however, being scheduled to finish the full experimental cycle in June 2006. Accordingly, this report serves as a progress report for the experiment from 1995 to 2000. The report is focussed on documenting the impacts of the experimental manipulations on some basic demographic parameters of the two primary target species, common coral trout (*Plectropomus leopardus*) and red throat emperor (*Lethrinus miniatus*) and the abundances of some of the prey species of coral trout. We present also an analysis of the experimental data with respect to estimating reef-specific biomass, experimental depletions and natural mortality of coral trout. Several other aspects of results from the ELF Experiment have been reported elsewhere (Adams 1996, 2002, Adams *et al.* 2001, Bean *et al.* in press, Williams *et al.* in press, Welch 2001) or are the subject of PhD theses and other reports in progress (A. Williams, James Cook University (JCU); M. Bergenius, JCU; R. Marriott, JCU; J. Mosse, JCU; R. Pears, JCU; C. Davies, Fisheries Research and Development Corporation (FRDC) Project 98-131) and will not be repeated here.

Materials and Methods

Experimental Design

The experiment involves 24 reefs spanning 7° of latitude or approximately half of the length of the GBR. The 24 reefs are grouped into four “clusters”, each with six adjacent reefs, located in four regions of the GBR: around Lizard Island (~14.5° S), off Townsville (~18.5° S), off Mackay (~20.5° S), and around Storm Cay (~21.5° S) (Fig. 5), in the north-western Swains reefs. Four reefs in each cluster had been zoned as Marine National Park B (MNP-B, ‘green’), meaning no fishing was allowed, for 10-12 years prior to the start of the experiment in 1995 and two reefs had been open to fishing historically (General Use zoning, GU, ‘blue’ reefs). The reefs included in the experiment (Table 1) were selected on the basis of the juxtaposition of the required set of MNP-B and GU reefs and extensive consultation with commercial and recreational fishers, scientists, and the GBRMPA³.

Figure 5: Map indicating the locations and arrangements of reefs used for the Effects of Line Fishing Experiment. Named reefs (except for Lizard Island) are those used in the ELF Experiment, and their treatments are given in Table 1.



³ Consultation was via a folded A3 pamphlet comprising maps of all potential reefs and a questionnaire. Over 4,000 pamphlets were distributed to individuals, community groups, and organisations on the Queensland Coast.

Table 1: Reefs included in the ELF Experiment, the Region in which they occur, the Latitude and Longitude (**Lat.**, **Long.**) at the centre of each cluster, their **Treatment** (with abbreviation), and the year in which they were fished intensively (if at all). Treatments comprised application of two processes, zoning and fishing, in different combinations in different years. MNP-B reefs designated for fishing in 1999 were effectively additional closed control reefs prior to being opened for fishing. **Treatment Abbreviations:** **MNP-C** – Marine National Park Control reefs (closed to fishing), **GU-F** – General Use reefs hard fished (ambient historical plus increased experimental fishing); **MNP-F** – Marine National Park Fished reefs (Pulse fished for one year).

Region / Lat, Long.	Treatment	Reef	Fished
Lizard 14°50'S, 145°30'E	MNP-Control (MNP-C)	MacGillivray Reef	Not
		South Direction Island	Not
	MNP-Fished (MNP-F)	Rocky Islets A	1997
		Eyrie Reef	1999
	GU-Fished (GU-F)	Rocky Islets B	1997
		14-133	1999
Townsville 18°30'S, 147°30'E	MNP-Control (MNP-C)	Dip Reef	Not
		Glow Reef	Not
	MNP-Fished (MNP-F)	Yankee Reef	1997
		Faraday Reef	1999
	GU-Fished (GU-F)	Knife Reef	1997
		Fork Reef	1999
Mackay 20°25'S, 150°15'E	MNP-Control (MNP-C)	20-137	Not
		20-142	Not
	MNP-Fished (MNP-F)	20-136	1997
		Bax Reef	1999
	GU-Fished (GU-F)	Liff Reef	1997
		Boulton Reef	1999
Storm Cay 21°25'S, 151°15'E	MNP-Control (MNP-C)	21-131	Not
		21-132	Not
	MNP-Fished (MNP-F)	21-130	1997
		21-133	1999
	GU-Fished (GU-F)	21-124	1997
		21-139	1999

Two of the MNP-B reefs in each cluster remain closed to all fishing other than annual research catch surveys. These reefs provide measures of natural fluctuations in reef fish populations under little or no impact of fishing, effectively acting as a closed control treatment. The remaining two MNP-B reefs in each cluster were opened to “at will” bottom line fishing for one year, after which they reverted to their former (MNP-B) status. One of these reefs in each region was opened between March 29, 1997 and March 28, 1998, whilst the second was opened between March 6, 1999 and March 5, 2000⁴. As much fishing as possible without incentive schemes was applied during the 12 months of pulse fishing on the

⁴ The second set of manipulations originally was intended to commence in 1998. These manipulations were deferred for a year to allow formal review of the merit of further opening of MNP-B reefs and because of persistent widespread changes (both increases and decreases) in catch rates of target species following the presence of a large tropical cyclone (Cyclone Justin, 1000km across) off the Queensland Coast in March 1997 and the onset of a strong ENSO event in March-April 1997. The cyclone / ENSO events coincided almost exactly with the beginning of the first set of manipulations and were followed by unusually early and large drops in water temperatures over much of the GBR. The exact nature of any causal relationship between these environmental events and changes in fishery-related data remains unclear.

experimental reefs by commercial, charter, and recreational fishers. On completion of each year of pulse fishing, each reef reverted to its 'closed' status indefinitely. Increased pulses of fishing also were encouraged on one of the two GU reefs in each cluster during each of these two years, after which the reefs were closed to further fishing for a period of five years. These GU reefs will revert to their 'open' status on March 29, 2003 (for those closed 1998) and March 6, 2005 (for those closed in 2000). The experimental design is schematised in Table 2.

Table 2: Summary of the experimental design and schedule of experimental treatments for one cluster of six reefs involved in the ELF Experiment. The experimental schedule is the same for each of the four clusters involved in the experiment. **Key:** C - closed to fishing; O - open to fishing; **P** – pulsed fishing; subscripts indicate the reefs treated first and second in each cluster.

Reef	Year										
	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
MNP-Control ₁	C	C	C	C	C	C	C	C	C	C	C
MNP-Control ₂	C	C	C	C	C	C	C	C	C	C	C
MNP-Fished ₁	C	C	P	C	C	C	C	C	C	C	C
MNP-Fished ₂	C	C	C	C	P	C	C	C	C	C	C
GU-Fished ₁	O	O	P	C	C	C	C	C	O	O	O
GU-Fished ₂	O	O	O	O	P	C	C	C	C	C	O

The geographic scope of the experiment and replication of treatment reefs within each region provides data from which to estimate regional variation in the biological characteristics of several main and secondary target species and regional variations in the responses of populations of those species to changed fishing pressure. Such regional considerations were considered essential for the application of Management Strategy Evaluations (MSE) to the entire GBR, especially given the expected high level of among-reef and regional variation in population dynamics of target species and the known regional variation in bio-physical structure of the environment (Doherty 1987, Maxwell 1968, Oliver *et al.* 1995, Williams 1991, Wolanski 1994) and properties of the Reef Line Fishery (Blamey and Hundloe 1993, Green *et al.* in press, Mapstone *et al.* 1996a, Trainor 1991).

Applying treatments in different years allowed the inclusion in results of inter-annual variation in treatment effects (over all regions). Explicit inclusion of inter-annual variation, available only with replication of treatments over years, was considered important because of the considerable inter-annual variation in recruitment of reef fishes (Doherty 1987, 1991, Doherty and Williams 1988) and the potential for events in any one year to influence substantially the outcomes of the experiment (Mapstone *et al.* 1996e, 1998a, Walters *et al.* 1988, Walters and Sainsbury 1990). Because only one reef of each zoning history (MNP-B, GU) in each region was subject to the pulse fishing treatment in each year, however, the interaction between treatment, year, and region is not separable from reef-reef variation within years and regions. This design compromise was adopted for two reasons: i) Including within-year replication of treatment reefs in each region would have increased the number of experimental reefs to at least 40 and would have proved logistically, environmentally, politically and economically difficult or unfeasible; and ii) this level of detail (Year x Region x Treatment interactions) is of little interest, given that management strategies for the GBR are unlikely to be tailored to specific inter-annual or regional changes in conditions but will be applied over several years at a time.

Field Sampling

All 24 experimental reefs (including controls) are sampled by structured underwater visual surveys (UVS) and line fishing catch surveys in the Austral Spring of each year. Surveys have been done from and including 1995. Sampling currently is funded until the Austral

Spring in 2005. Surveys during 1995 and 1996 provided 'baseline' (with respect to the proposed manipulations) data for the reefs prior to commencing manipulative treatments. Surveys during 1997-98 and 1999-2000 provided estimates of the effects of the increases in fishing pressure, and surveys following reef closures provide data on the rebuilding of stocks following depletion by fishing.

Additional UVS and catch surveys of each of the two manipulated reefs (one MNP-Fished and one GU-Fished reef) and one un-changed (control) MNP-B⁵ reef in each region were done to monitor changes during the years of manipulations. This meant that additional visual surveys were planned for March 1997 and March 1999, immediately prior to beginning manipulations, and August 1997 and August 1999, between the Autumn and Spring surveys. Additional catch surveys of the same three reefs in each cluster were planned for March, May and August in both 1997 and 1999 and April of 1998 and 2000, immediately after closure of the manipulated reefs. Thus, the two manipulated reefs and one control reef in each cluster were to be sampled 5 times during the manipulations, including surveys immediately prior to and following increased fishing pressure. In practice, the surveys immediately prior to the reef openings in 1997 were impossible because of bad weather throughout March 1997 caused by Cyclone Justin. Data from these surveys of only 12 of the 24 reefs (three per region) are used primarily for the derivation of depletion estimators of reef-specific biomass and are not otherwise considered in this report.

Line Fishing Catch Surveys

Line fishing catch surveys are done in the Austral Spring of each year to coincide with the spawning period of common coral trout (*Plectropomus leopardus*). The spring surveys are designed to sample fish during their spawning period when gonads are active and provide most information about the reproductive status of local populations. The surveys are timed around the full moons, however, to avoid the times of peak spawning activity (centred on Spring new moons, Samoily 1997) and avoid potential biases incurred from sampling fish when they are aggregated for spawning⁶.

Catch surveys are done via the charter of an active commercial fishing operation with master fisherman and four dory-fishermen, accompanied for the surveys by four research staff. Whenever possible, at least some of the dorymen were constant among surveys. The high turnover of crew among vessels in the reef line fishing fleet, however, meant that considerable changes in dorymen occurred among surveys. The same master fisherman was used for all surveys between 1995 and 2000, but different master fishermen had to be employed thereafter.

Fishing gear is standardised among fishers and over time to be comparable with standard contemporary gear used in the commercial RLF on the GBR. We use 80lb monofilament fishing line with a "running sinker" rig, consisting of a bean sinker rigged on the main line above a single 8/0 hook (Mustard pattern 4279). Western Australian pilchards are used as the standard bait, as is the case in the commercial RLF. The use of "hard bait" (strips of freshly caught fish fillet) is not permitted, again to standardise methods among fishers and over time. Each fisher uses only one line at any time.

Sampling at each reef involves the same general design, as follows. Each reef is divided into six approximately equal sized, contiguous 'blocks', three on the windward aspect of the reef and three on the leeward aspect (Fig. 6). Block boundaries are located by GPS during

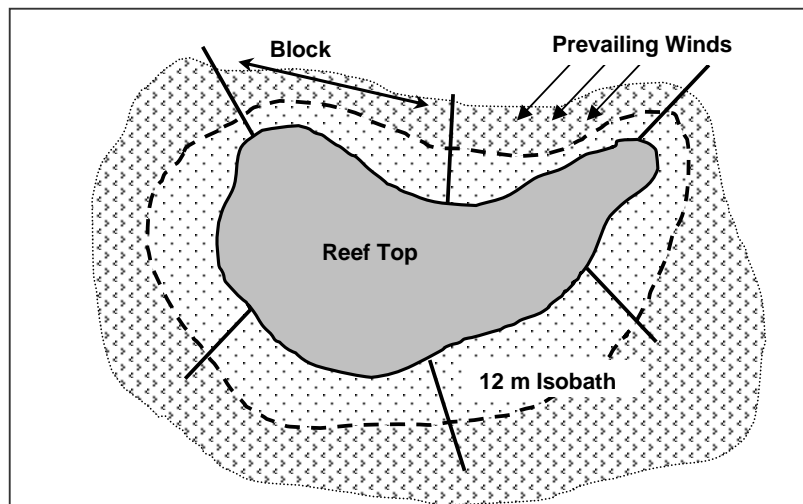
⁵ During the first set of manipulations (1997), the MN-P B reefs that were destined to be opened to fishing in 1999 were used as these controls. This strategy was adopted to avoid impacts of additional sampling during 1997-98 on the overall control reefs, effectively limiting impacts of frequent sampling to those MNP-B reefs that would eventually be opened to fishing anyway.

⁶ Studies of the spawning behaviour of *P. leopardus* indicates that for 3-4 days either side of the Spring new moons they aggregate daily to spawn (at dusk) but disperse between new moons, and sometimes disperse each day after spawning within the spawning period (Samoily and Squire 1994, Samoily 1997, Zeller 1998).

each survey. Each reef is sampled on a single day on each sampling occasion⁷, with sampling being stratified into morning (AM) and afternoon (PM) 'sessions' of 4-5 hours duration. One commercial dory-fisherman is allocated randomly to each of the three most up-stream blocks (as dictated by tidal flow⁸) in the AM session of each sampling day, and then again to one of the remaining three blocks in the PM session of that day. A fourth doryman 'roves' over all three blocks being fished in each session.

Each doryman is required to distribute their fishing effort approximately evenly over the block, doing at least four 'hangs' (defined as a single location where a fisher sets anchor and commences to fish) in water less than 12m deep to correspond with the limitations of Underwater Visual Surveys (UVS) and four hangs in depths of 12 ~ over 30m to extend the range of data beyond the limitations of UVS and out to the margin of reef habitat. Each hang is between 5 and 20 minutes duration. Hangs are separated by at least 100m.

Figure 6: Diagram illustrating sampling stratification at each reef. Underwater Visual Surveys (UVS) were done within the 12m isobath, whilst Catch Surveys (CS) sampled equally in both depth strata (> and < 12m). Lines extending from reef perimeter indicate boundaries of 'blocks' within which UVS transects and CS 'hangs' were distributed. **Key:** - - - 12m Isobath; – Margin of reef habitat.



Each doryman is accompanied by a researcher who ensures that the designated sampling regime is adhered to and records a range of information about fishing locations (GPS location, depth, arrival and departure times), fishing practice (searching and anchoring times, line-reset times, time of each retrieval and result – e.g., no bait, species caught, etc), and characteristics of the catch (species, length). All fish caught up to the quota limits imposed

⁷ In Spring 1995, each reef was sampled on two days, about 1 week apart. One day involved the structured fishing described here, whilst the second involved 'at will' fishing, monitored by researchers in each dory. This work was done to compare the catch characteristics and fishing behaviour of our structured surveys with 'normal' fishing conditions in the commercial fleet. This report uses only data from the structured fishing.

⁸ It is a common contention among experienced line fishers that the direction of "run" (tidal current) is a dominant determinant of catches at a particular location (Davies 1995a,b). This is most likely related to the feeding behaviour of the target species and their interaction with the gear. Thus, catches are generally higher when fishing the "run on" side of a reef, where the prevailing current is flowing onto and over the reef, than the "run off" side, where the water flows off the reef into deeper water. We stratified sampling effort by tidal flow to standardise for this potential effect, wherever possible fishing each block during the "run on" phase of the tidal cycle for that block.

by GBRMPA and QFMA / QFS permit conditions⁹ are tagged and kept for processing that evening. Fishing continues for the full day on each reef irrespective of whether the quotas have been reached, with any fish caught over-quota being measured and held in live fish tanks in the dories and then released at the end of each hang.

In the evening of each day, the catch is processed for sale¹⁰ and the collection of biological samples. Each fish is identified (independently of its identification in the dory), weighed, has gonads and otoliths removed, labelled and stored and is then filleted or gilled and gutted before being snap-frozen for sale. Most fish are processed completely at sea, but where this cannot be done, 'frames' (head, viscera and skeleton) are frozen and processed on return to the laboratory.

Underwater Visual Surveys

Underwater Visual Surveys (UVS) were done twice annually, once in Autumn (April-May) and once in Spring (September-November/December) until 2000 and once annually, in Spring, thereafter. The Autumn surveys were timed such that young-of-the-year coral trout could be distinguished by size from older fish, hence providing an index of recruitment to the reef following larval dispersal. These Autumn surveys were discontinued because a) counts of very small *P. leopardus* were insufficient to be informative, b) there was no clear differentiation of a mode in the distribution of estimated sizes (Ayling 1983) that could be interpreted to represent the young of the year cohort, and c) the additional data did not alter substantially the inferences drawn from only Spring surveys and so were considered an extra cost offering only marginal additional benefit.

Catch and underwater surveys are timed closely within each sampling period, such that UV surveys precede catch surveys of each reef by a few days. Sampling within each reef is stratified similarly by blocks for both underwater and catch surveys. Underwater visual surveys involve a team of 3 divers and a boat person and are based from a chartered support vessel. The same observer counts all fishes in all surveys¹¹, whilst various assistant divers count various benthic organisms. Diving safety considerations mean that these counts are restricted to the shallow depth stratum, between 2m and 12m depth. Each reef (6 blocks, 5 transects per block) takes one day to sample on each occasion.

Transects are laid and counts done following the methods recommended by Mapstone and Ayling (1998) and used by Mapstone *et al.* (1998b,c, 1999) and Ayling and Ayling (1992b, 1994, 1995). One diver runs out a 50 m fiberglass tape along the reef slope at a depth of about 2-8 m. The principal observer swims beside the tape layer, counting each species of coral trout, lethrinids, lutjanids and (where feasible) chaetodontids, and scarids and caesionids as multi-species groups within an estimated 5 m of the seaward side of the tape. Scarids are counted in two size classes (summed across species): 8-25cm total length (TL), and > 25cm TL, corresponding to whether they were likely to be within or above the size preyed upon by coral trout (St John 1995). Caesionids are summed (over all species) separately for each 10m segment of the 5m wide transect (i.e., five subdivisions) and then summed for the transect. Bait fish schools (hardy heads and clupeids) are recorded as present (1) or absent (0) within 10m of each side of each 10m segment of transect, giving a bait fish index of 0-5 for each transect. *Acanthaster planci* are also counted within the 5m wide transects.

⁹ Total allowable catch was 300 *P. leopardus* per reef per survey, with up to 90 undersized *P. leopardus* (<38cm TL), and up to 350 individuals of all other species combined, also with various species-specific limits on the numbers of sub-legal sized fish that could be retained.

¹⁰ Following extraction of biological samples, all catch was processed on the main vessel as in the commercial fishery and product from legal-sized fish sold, with the proceeds of the sale returned to the ELF Project to partially off-set the costs of chartering the fishing operation for the catch surveys.

¹¹ Duplicate counts are done concurrently by a back-up observer once per year, with indexing of the observers' counts against each other, to cater for any future absence of the primary observer.

When the principal observer completes the large transect, he returns along the tape counting pomacentrid fishes 0.5 m each side of the tape (50 x 1m). The pomacentrids are counted in four categories, corresponding to their relative importance in the diet of coral trout: *Amblyglyphidodon curacao*, *Chrysiptera rollandi*, *Pomacentrus moluccensis*, and the remainder of pomacentrids (grouped). The tape layer follows, winding in the tape and summing hard coral intercepts for the first 20 m of the return, soft coral intercepts for the next 20 m of the tape, and sponge intercepts for the last 10 m (back from the start point). These techniques maximise consistency and minimise potential for errors as the principal observer is responsible for recording all the most important groups.

At the start of each transect the tape is run out at right angles to the proposed transect line to show the principal observer the 5m width of the transect. At the end of the first pass along the transect, the principal observer indicates his estimate of the width of the transect and this is measured with another tape by the tape layer and recorded.

The minimum total length of fish recorded in the counts is 6 cm for *Plectropomus* spp, 10 cm for lethrinids and lutjanids, 4 cm for chaetodontids and 2 cm for pomacentrids. Total lengths are estimated for all *Plectropomus* spp and selected lethrinid and lutjanid species harvested by the RLF. Length estimation re-training using wooden coral trout models is undertaken by the principal observer at the beginning of each survey. All data are recorded on ruled waterproof paper sheets in a standard format that has been well trialled in previous work and makes for easy computer entry.

Laboratory Methods

All sagittal otoliths and gonads are catalogued and stored for later processing as soon as possible following their return from field surveys. Ages of fish are estimated from counts of annuli on otoliths and reproductive stages of fish are assessed from histology of the gonads. Samples are processed roughly in order of species' abundances and economic importance to the fishery. Accordingly, the main focus of work to date has been on the common coral trout (*Plectropomus leopardus*) and the red throat emperor (*Lethrinus miniatus*). Only these harvested species are discussed in detail in this report. Results of reproductive analyses are the subject of other reports arising from the ELF Experiment, including Adams (1996, 2002), Adams *et al.* (2001), Bean *et al.* (in press) and several PhD theses and a companion FRDC project report (FRC Project 98-131) and are not discussed here.

Otolith Processing

The right sagittal otolith is used to estimate the age of all fish unless it is missing or damaged, in which case the left otolith is used. The right otolith is used for age determination simply for consistency, there being no difference between left and right otoliths in either weight (Hatcher 1996) or counts of annuli (Ferreira and Russ 1994).

Methods of estimating age from otoliths differed among species, depending on the characteristics of the otoliths. Otoliths of red throat emperor and a number of other lethrinids are remarkably translucent and initial trials showed that reading these fishes' otoliths whole provided the same age estimates as the sectioned otoliths. Accordingly, all such otoliths were read whole. Otoliths from all other species were read in sectioned form, as described below. The reading procedures and rules for assigning age were the same for all species, irrespective of how the otoliths were prepared for reading.

Each sagittal otolith is weighed to the nearest 0.1 mg using a Sartorius electronic balance, before being embedded in epoxy resin, and then sectioned transversely through the core with a Multi-Drive low-speed saw with diamond edged blade. Sections are mounted on glass slides with Crystal Bond 509 adhesive, ground on 600- and 1200-grade carborundum paper, and polished with 0.3 µm alumina micropolish before reading.

Annuli are counted in sectioned otoliths under a dissecting microscope at 40X magnification with reflected light and a black background, following the methodology described by Ferreira

and Russ (1994). Annuli are counted from the nucleus to the proximal surface of the sagitta, along the ventral margin of the sulcus acousticus. The outermost annulus (opaque band) is counted providing that the otolith has a hyaline outer margin.

Otoliths from each survey are read in strictly random order with no prior knowledge of location of capture or size of the fish from which each otolith came. Each otolith is read by at least two independent, experienced readers. If the counts from the two readers are the same, it is accepted as the age of the fish. If the counts differ, a third count is made by a third independent, experienced reader. If the third count matched either the first or second count, that age is accepted. If there is no agreement between any two of the three readers, a final count is made by all three readers together. If no agreement can be reached on this final count, the otolith is discarded.

Estimates of Catch and Effort from Experimental Reefs

Data from which to estimate the effectiveness of the experimental manipulations in changing effort and catch on the experimental reefs were collected from three independent regimes. First, daily catch and effort were estimated from compulsory commercial fishery logbook data. Second, aerial and vessel surveillance data provided intermittent estimates of (potential) fishing activities on experimental reefs. Third, researchers were placed on site at MNP-Fished reefs when they were first opened to fishing.

Compulsory Fishing Logbooks

All Queensland commercial fishermen have been required by QFMA and QFS since 1988 to report their fishing location and catch (by species or species groups) for each day. Data must be reported at the scale of 30' x 30' grids and, since 1993, optionally at the scale of 6' x 6' sites within these grids. Fishers also are expected to report the numbers of crew, lines and dories used each day, though these data are optional. Complete data from the commercial RLF from 1988 onwards were made available for this project by the QFMA and QFS. Charter vessel operators also have been required to keep logbooks of their fishing activities since 1996, though reporting coverage was not complete until 1997-98¹².

Aerial and Vessel Surveillance

The GBRMPA requested, on behalf of the ELF Project, that existing aerial surveillance flights by the Queensland Department of Environment (QDoE) and Coastwatch over the GBR be diverted and/or increased where possible to survey experimental reefs immediately prior to them opening and subsequently. This would have resulted in planned over-flights, on average, approximately twice per week. In practice, however, this frequency was reduced because of funding constraints (QDoE) and bad weather (preventing flying) or higher priority demands on aerial surveillance (e.g., customs activities). Over-flight frequencies during 1997 ranged from approximately 1.5 per week for the Lizard Island Cluster to slightly more than one flight per fortnight over the Mackay and Storm Cay Clusters, and have become considerably less frequent since. On each flight the position of each vessel sighted, its type and apparent activity, and, where possible, registration details were recorded. In practice, the frequency and accuracy of records of "line fishing" vessels proved to be insufficient for estimation of total effort and are not considered further.

On-site Observations

Researchers on-site at the four MNP Fished reefs being opened in each year made continuous observations of fishing activity (effort and catch) during the initial weeks that the reefs were opened to fishing. Each researcher worked from a 6m Yamaha Southwind utility boat that allowed them to circumnavigate the treatment reefs and visit all fishing operations present. Researchers 'camped' on islands (in the Lizard region), on fishing vessels or

¹² Note that because the charter vessel logbook programme is new there is likely to be less certainty about the data arising from it than from other sources.

Queensland Boating and Fishing Patrol vessels each night and remained on-site until there were no vessels present on which to stay. Where intermittent fishing activity persisted above 'normal' levels, on-site observations were prolonged by chartering vessels to accommodate observers or accompanying fishing operations that were working in the vicinity of experimental reefs. On-site observations ranged from 7 to 23 days after opening reefs.

Data Management

All data from the ELF Experiment (and other aspects of the ELF Project) are stored on centralised relational databases on Windows NT Servers. The databases are being developed for long term integrated management of the data. Links with other relevant databases (e.g., long-term monitoring data from AIMS, QFS logbook data) have been anticipated in the structure of the database.

All data collected by the ELF team are entered twice into separate dedicated Microsoft Access databases via data-entry forms with filters to trap suspect (e.g., out of reasonable range) data. The two data entry processes are independent with all data entered by different operators on each occasion. These 'double punched' data are then compared with purpose built software, disparate values compared with original datasheets and errors corrected in either (or both) databases. Finally, a wide range of error checking (e.g., random data audits, range checks, verification of sampling sizes and times, etc.) is done prior to analyses to trap inconsistencies or suspect data that survive the prior error checks.

Analytical Methods

Preliminary Processing of Data

Research Data

A range of variables were derived from field and laboratory data prior to analysis. Underwater Visual Survey count data were transformed to densities of fish per 250m² whilst the summed index data for each transect were analysed for estimates of baitfish abundance. Catch rates from catch surveys were calculated as numbers of fish caught per line hour (Catch per Unit of Effort, CPUE) per block per survey. These indices were calculated by summing the numbers of fish caught (the catch) and actual fishing times (*i.e.*, time between the first bait being set and the last bait being retrieved, the effort) across all hangs within each block and dividing the catch by the effort. Mean fork lengths (FL) and ages of fish were calculated across all fish caught and processed within each block and survey. Total rates of mortality were estimated from the catches each of several cohorts over consecutive years. Estimates were computed for data from GU reefs in all years (total mortality) and MNP reefs in years before they were opened to fishing (Natural mortality). These analyses included only fish at and older than ages expected to be fully recruited to the fishery and available to the gear (four years and older for *P. leopardus*, six years and older for *L. miniatus*) and from cohorts that were represented in catches from at least three successive years. These cohort-specific estimates of mortality were then averaged within reefs to give average estimates of mortality on MNP Closed reefs and GU Fished reefs in each Region. Mortality was compared among Zones and Regions by 2-factor ANOVAs with the fixed factors Region and Zone and reef-specific estimates of mean total mortality as replicates.

If fishing times were missing because either the 'start fishing' or 'end fishing' variables were absent, fishing time was derived from substituting the nearest (in time) available datum for the missing value. The substitutions for missing 'start fishing times' were sought from (in order) 'set time', 'end search time', 'start search time' time of first fish caught' for the same hang and 'end fishing time' from the previous hang. Substitutions for missing 'end fishing times' were sought from (in order) 'time of last fish caught' in the same hang and 'begin search time', 'end search time', 'set time' and 'start fishing time' for the following hang. If none of these substitutions were not available (e.g., because of failure to record times in dories), then the fishing time for the hang was derived from the fishing times for other hangs in the same block by taking a random normal variate with mean and variance of the recorded

fishing times for other hangs in the same block or (if few other times were available) other blocks at the same reef on the same survey.

Finally, there was one case where data were missing from an entire reef in the Lizard region because bad weather prevented sampling that reef during the spring survey in 1996. In this case, the block-level data were estimated as a random normal variate with weighted mean and variance of the corresponding variables recorded from the same block in the one previous and three following years.

Logbook Catch and Effort Data

Catch and effort for the experimental reefs could only be resolved from 6' x 6' site-level compulsory logbook data. Use of only site-level data, however, would mean ignoring much of the logbook data since some fishers report only at grid-scale (30' x 30'). To allow use of the data reported only at grid-scale, we used the available site-level data to determine the relative distribution of fishing effort and catch among sites within each grid within each year, or averaged over all years when necessary, and used these distributions to allocate grid-scale data among sites within grids. This has the advantage that all reported catch and effort is used, but the disadvantage that some catch may be attributed to sites from which it was not taken. An underlying assumption in this process is that when reporting only at grid level, fishers do not allocate their effort significantly differently than when reporting at site level. Effort was calculated as both fishing operation-days (an operation comprising a main boat and its dories and crew) and as estimated line-days. Catch was calculated as whole weights of fish.

Because catch and effort data are not reported by reef, we could only estimate reef-specific catches by attributing catch or effort within a six minute site among reefs that existed in that site. We did this pro-rata according to the perimeter of reefs within each site. The zoning status of the reefs was used to weight the distribution of catch and effort as follows. Where only MNP (closed) or only GU (open) reefs were present in a site, all catch and effort from the site was attributed to reefs strictly pro-rata by their perimeters. Where both MNP and GU reefs were present, the allocation of catch and effort was weighted such that reefs closed to fishing were attributed 20% of that catch and effort that would have been appropriate based on their perimeters and the remaining catch and effort was attributed to the open reefs. In these ways, we allowed for inadvertent or deliberate infringements of closures, but assumed that such infringements would not equate with normal levels of fishing on open reefs.

Catch and effort data for the species or groups 'coral trout'¹³, red throat emperor (*L. miniatus*), and 'other mixed demersal reef species' were summed for each 6'x6' site over 12 month intervals defined by the start and end dates of manipulations. That is, for reefs that were pulse fished between March 29 1997 and March 28, 1998, data were aggregated from March 29 in one year to March 28 in the following year. For reefs that were pulse fished between March 6 1999 and March 5, 2000, data were aggregated from March 6 in one year to March 5 in the following year. Data from MNP Control reefs were aggregated from March 6 to March 5 because it was expected that the major pulse in effort and catch would occur in the first months of manipulations and would be relatively trivial at the end of the manipulation periods. Thus, the March 6-March 5 period would capture such pulses (on fished reefs) for both manipulative periods and provide for best comparison with all manipulation reefs.

Catch and effort were summed within reefs and divided by the reef perimeter to estimate effective catch (or effort) density and facilitate meaningful averaging of catch and effort statistics across reefs with similar zoning and treatment status in each year but with considerably different sizes. Catch density on reefs subject to pulse fishing in 1997 was compared with catch density on non-pulsed, but otherwise similar, reefs in the same year

¹³ The generic term 'coral trout' is used here because there is no discrimination in logbook data among species of coral trout. *P. leopardus*, *P. laevis*, *P. maculatus*, *P. aereolatus* and *Variola* spp. all occur in the catch but are reported collectively with *P. leopardus* as 'coral trout' and rarely reported separately).

and all similarly zoned reefs in previous years as the basis for assessing the effectiveness of intended fishing or closure treatments.

Tests for Effects of Fishing, Region and Zone

Only selected research variables were analysed for this report. Variables were selected according to their data density across all reefs and to illustrate different characteristics (abundance, age, size) of patterns in the demographics of harvested species or their prey. Accordingly, the following variables were analysed independently for this report: Population density (from UVS) and CPUE, mean age and mean fork length (FL) (from Catch Surveys) of legal sized and under sized (or sub-legal) *P. leopardus* (hereafter coral trout) and legal sized *L. miniatus* (hereafter red throat emperor); and population densities of *A. curacao*, *C. rolandi*, *P. moluccensis*, ‘other’ pomacentrids, caesionids, small ($\leq 25\text{cm}$) and large ($> 25\text{cm}$) scarids. Each of these variables was analysed in the same way, as described below. Red throat emperor are caught rarely north of Cairns and analyses for this species included data only from the southern three regions, Townsville, Mackay and Storm Cay.

Prior to analysis, reef means were derived as weighted means of block means, where the weights for each block were the numbers of samples (e.g., hangs, fish) from that block. Reef means were used in analyses because reefs were the units of experimental manipulation and so constituted the true replicates for comparisons of treatment, zoning or regional effects. This process also had the advantage that the data subject to analysis, being means of many other data, were normally distributed (by the Central Limit Theorem) and also were generally homoscedastic. Accordingly, data were not transformed before analyses. Data within reefs were thus sub-samples of the experimental units collected in stratified ways to adequately represent the overall status of each reef and were not of interest per-se for this report. Reef means were compared among Regions, Zones and Treatments over survey Years by a series of repeated measures analyses of variance run using SAS software.

The repeated measures analyses involved the between-subject (subjects = reefs) factors Region (Lizard, Townsville, Mackay, Storm Cay), Zone (GU, MNP-B) and Treatment (MNP-Control, MNP-Fished, GU-Fished) nested within Zone. Within-subject (repeated measures) factors were either calendar Year (1995-2000) or Treatment Year (Baseline years 1 and 2, year of Pulse fishing, or the first year of post-treatment stock Rebuilding), dependent on the model involved (below). All factors were considered fixed effects.

The levels of the factor Treatment Year were constructed from considering time in relation to the pulse fishing events for each reef, as follows. The ‘Baseline 1’ and ‘Baseline 2’ treatment years were comprised of data from 1995 and 1996 respectively for all treatments (MNP-Control, MNP-Fished and GU-Fished) in all regions. The ‘Pulse’ year comprised data from the MNP-Fished and GU-Fished reefs in each region from the year in which each reef was subject to pulse fishing. That is, one MNP-Fished and one GU-Fished reef in each region would be represented by 1997 data whilst the other reef in each of these treatments would be represented by data from 1999. Similarly, the ‘Rebuild 1’ year comprised data from 1998 or 2000 for reefs that were fished in 1997 or 1999 respectively. Finally, the MNP-Control reefs were represented in the ‘Pulse’ Treatment Year by the average of data from 1997 and 1999 (the years in which fishing treatments were applied to other reefs) and represented in the ‘Rebuild 1’ Treatment Year by the average of data from 1998 and 2000, being the years following closure of fished reefs.

Two primary sets of analyses were used to explore different questions. First, the effects of Zone and Region were examined by three applications of the following model:

$$y_{ijkl} = \mu + R_{i...} + Z_{j..} + RZ_{ij..} + \varepsilon_{k(ij)} + Y_{...l} + RY_{i..l} + ZY_{j..l} + RZY_{ij..l} + \xi_{k(ij)l} \quad (1)$$

where:

- y_{ijkl} is the observation from reef k in Zone j and Region i in Year l ;
- μ is the true population mean for variable y ;
- $R_{i...}$ is the effect of being in Region i (Lizard, Townsville, Mackay, Storm Cay);
- $Z_{j..}$ is the effect of Zone j (Marine National Park-B, General Use);

$\varepsilon_{k(ij)}$ is the residual variation attributable to being on reef k in Region i and Zone j ;
 $Y_{...l}$ is the effect of Year l (1995-2000);
 $\xi_{k(ij)l}$ is the interaction between reefs and Years; and
 The remaining terms are the interactions among the main effects R , Z , and Y .

The between-reefs residual Mean Square, $\varepsilon_{k(ij)}$, was the error variance for tests of the main effects of Region and Zone and their interactions, whilst the effects of Year its interactions with Region and Zone were tested against the within-reef residual Mean Square, $\xi_{k(ij)l}$.

The three applications of this model involved three different subsets of reefs and years: a) all reefs during the baseline years 1995 and 1996 only; b) three MNP reefs and one GU reef unaffected by pulse fishing in 1997 considered over the four years 1995-98; and c) two MNP Control reefs and the two General Use reefs compared over all years 1995-2000¹⁴. Effects of Zone were inferred from all three analyses, whilst the Regional patterns were inferred only from analyses a) and c).

Second, the impacts of the experimental treatments were explored via tests for selected terms in the following model:

$$y_{ijklm} = \mu + R_{i...} + Z_{j...} + RZ_{ij...} + T(Z)_{.k(j)..} + RT(Z)_{ik(j)..} + \varepsilon_{l(ijk)} + T-Y_{...m} + RT-Y_{i...m} + ZT-Y_{j...m} + RZT-Y_{ij...m} + T(Z)T-Y_{k(j).m} + RT(Z)T-Y_{ik(j).m} + \xi_{l(ijk)m} \quad (2)$$

where:

y_{ijklm} is the observation from reef l within Treatment k in Zone j and Region i in Treatment Year m ;
 μ is the true population mean for variable y ;
 $R_{i...}$ is the effect of being in Region i (Lizard Is., Townsville, Mackay, Storm Cay);
 $Z_{j...}$ is the effect of Zone j (Marine National Park-B, General Use);
 $T(Z)_{.k(j)..}$ is the effect of Treatment k (MNP Control, MNP Fished, GU Fished) in Zone j ;
 $\varepsilon_{l(ijk)}$ is the residual variation attributable to reef l ;
 $T-Y_{...m}$ is the effect of Treatment Year m (Baseline 1, Baseline 2, Pulse, Rebuild 1);
 $\xi_{l(ijk)m}$ is the interaction between reefs and Treatment Years; and
 The remaining terms are the interactions among the main effects R , Z , $T(Z)$ and $T-Y$.

The between-reefs residual Mean Square, $\varepsilon_{l(ijk)}$, was the error variance for tests of the main effects of Region, Zone and Treatment(Zone) and their interactions, whilst the effects of Treatment Year and its interactions with Region, Zone and Treatment(Zone) were tested against the within-reef residual Mean Square, $\xi_{l(ijk)m}$.

In this second suite of analyses, we were interested only in the interaction between Treatment within Zone (Treatment(Zone)) and Treatment Year, either alone or in interaction with Region and/or Zone. These interactions would indicate differences among Treatments in the response of variables over years, potentially in response to Pulse fishing, and would be a necessary pre-requisite to the inference of an effect of the fishing treatment as opposed to effects of Zone, Region or time unrelated to the fishing treatment. If one or more of these key interactions was statistically significant, the model was broken down into sub-models to explore the sources of these interactions. The sub models involved; a) tests of Treatment Year and Region effects for GU Fished Reefs alone; b) application of the above model to only MNP Control and MNP Fished Reefs; and c) tests of Treatment (MNP Control vs MNP Fished) and Region effects for MNP reefs only in the separate treatment events, 1997-98 and 1999-2000. In the latter analyses, the model involved the orthogonal factors Region, Treatment (between subjects effects) and calendar Year (within subjects).

Both multivariate and univariate tests of repeated measures were run in the above analyses [(1) and (2)], and the multivariate result accepted if the results were qualitatively different (e.g., one significant, the other not). The Huynh-Feldt correction for non-sphericity of variance-covariance error matrices was applied to univariate tests (Winer *et al.* 1992).

¹⁴ Complete age data for coral trout were available at the time of analyses only for the years 1995-99. Thus, only the Treatment Years Baseline 1 and 2 and Pulse and calendar years 1995-99 were analysed.

Terms were considered statistically significant and worth further exploration if $P(\text{data}|H_0) < 0.1$. Sums of Squares (SS) and degrees of freedom (df) of interaction terms were pooled with the relevant residual SS and df when $P > 0.25$. Tests and pooling progressed from the highest order interaction down to the two-way interactions with each interaction remaining in the model being tested after any term was 'pooled' to maximise the statistical power of tests at each step. Main effects were not considered in these pooling procedures. For brevity, test statistics and associated probabilities are reported only for significant effects.

Preliminary Applications of Depletion Estimators to ELF Experimental Data

Punt et al. (2001) explored the application of population dynamic models to ELF field data in order to estimate reef-specific biomass, natural mortality, reef-specific biomass depletions from fishing, catchability and the relative merits of Underwater Visual Surveys and Catch Surveys for providing abundance indices for coral trout. This work has been extended, but is still under development. The ability to estimate biomass precisely depends (*inter alia*) on the amount of contrast in the data, which will be maximised after the re-building phase of the ELF Experiment is completed. The methods to be used to estimate biomass and fishing mortality depend on the nature of the data and the properties of those are being refined as part of the experiment. Accordingly, we do not present definitive results from this work at this stage. We include in Appendix A, however, an updated description of the methods being pursued and results of a range of sensitivity analyses done with the data available so far. We do not report on this work other than in this appendix.

Results

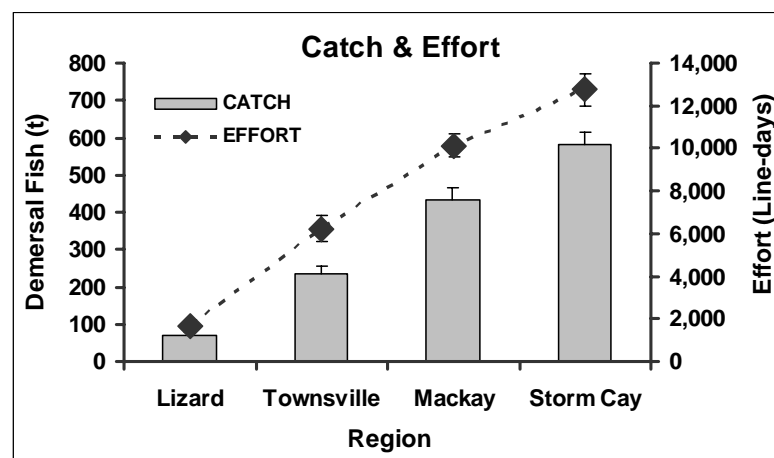
Catch and Effort on Experimental Reefs

On-site observations during the opening of the experimental reefs revealed that the bulk of the fishing effort and catch during the periods following opening of MNP Reefs to fishing was by commercial fishing operations (Table 3). The average annual catch and effort by commercial fishers in the degree of latitude around each set of experimental reefs estimated from compulsory logbooks varied regionally (Fig. 7), as would be expected from previous analyses of regional variation in commercial catch and effort (Mapstone *et al.* 1996a).

Table 3: Summary harvest and effort recorded by on-site observers during the initial 2 weeks after MNP reefs were opened to fishing in each region in 1997 and 1999.

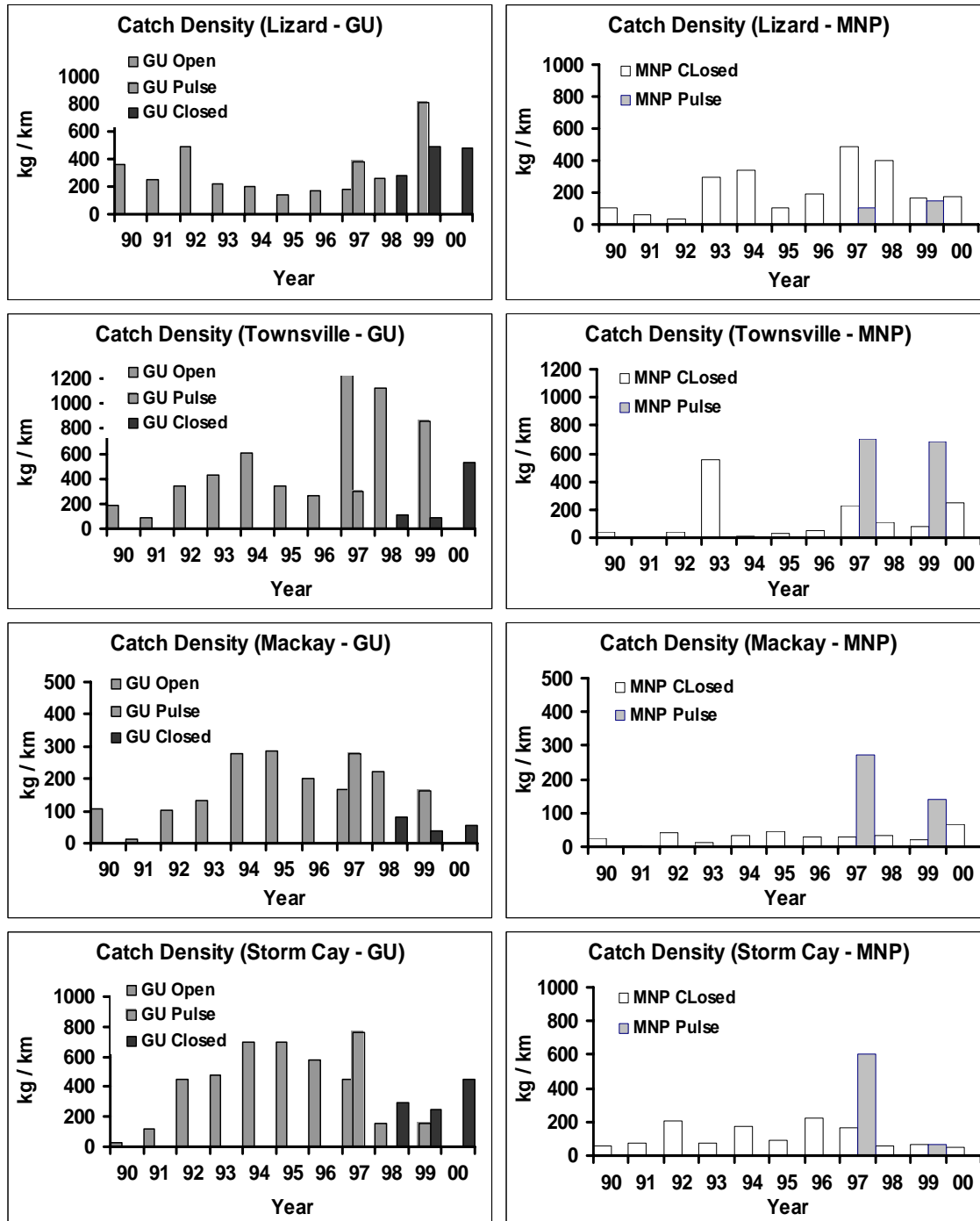
		N° Boats			Harvest (kg)	
1997						
Region	Comm.	Charter	Rec.	Coral Trout	Others	
Lizard	1	1	1	12	0	
Townsville	9	2	4	1217	1514	
Mackay	8	1	2	1724	1338	
Storm Cay	14+	0	0	2587	1912	
1999						
Region	Comm.	Charter	Rec.	Coral Trout	Others	
Lizard	3	1	2	467	333	
Townsville	5	2	1	279	145	
Mackay	10+	10	10	2925	2040	
Storm Cay	3	0	0	492	318	

Figure 7: Average annual catch of demersal species and commercial fishing effort between 1989 and 1997 in the 1° of latitude around each cluster of experimental reefs.



The effectiveness of experimental treatments varied among regions and zones, as did the historical amounts of fishing around reefs open and closed to fishing (Fig. 8). Catches attributed to MNP reefs in the Lizard region were broadly similar to those attributed to GU reefs and were not greatly elevated on treatment reefs during the pulse fishing years. Indeed, catch density around closed MNP reefs in 1997-98 and GU closed reefs in 1999-2000 were among the highest since 1990 (Fig. 8). The degree to which these elevated catch densities represent infringements of the closures or fishing around the margins of closures cannot be verified.

Figure 8: Mean annual catch density (kg fish per km reef perimeter) of all demersal species from ELF Experimental reefs zoned General Use B (**GU**, Left) or Marine National Park B¹⁵ (**MNP**, Right) in each region and year since 1990. Data were averaged over reefs within zones having the same status (Closed or Open) or subject to the same experimental treatment (Pulsed) in each year. The effectiveness of attempts to increase fishing are inferred from comparing reefs 'Pulsed' in 1997 or 1999 with similar but un-manipulated reefs in the same or previous years.



¹⁵ Because catch and effort data are reported by, at best, 6'x6' site, legitimate catch and effort records exist for sites containing one or more closed reefs but also containing area open to fishing. In the above figures, we have assigned such catch to the closed reef as described in the methods.

Opening MNP-Fished reefs to fishing resulted in dramatic increases in catch density around those reefs in the remaining three regions, though the effect was absent from the Storm Cay region in 1999 (Fig. 8). Evidence was ambiguous as to whether attempts to increase fishing on selected GU reefs in 1997 and 1999 were effective in any region. Although sharp increases in catch density were evident on open reefs off Townsville, in 1997 and 1998 these increases were on the GU reef not intended for increased fishing. Catch density from the GU treatment reef off Townsville in 1999 was significantly greater than in years prior to commencing manipulations, though not as great as on the same reef in the previous two years (Fig. 8). Catch densities on GU treatment reefs in the Mackay and Storm Cay regions during 1997 were slightly greater than on the non-treatment GU reefs, but in 1999 catch densities on GU treatment reefs were among the lowest recorded since 1990 (Fig. 8).

Tests for Region, Zone and Effects of Fishing.

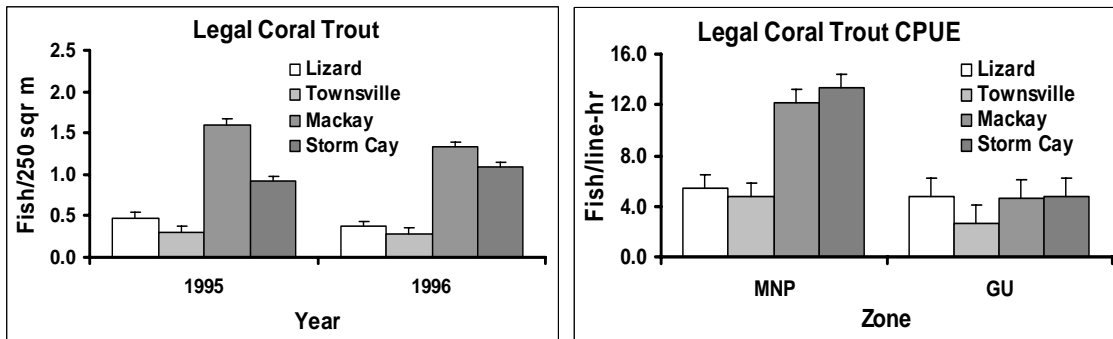
Patterns Among Regions

Regional patterns were common in the analyses across both target and non-target species. Though regional patterns in abundances were variable among years, in most cases abundances of coral trout, red throat emperor, most pomacentrids and caesionids were considerably greater in the southern regions of Mackay and Storm Cay than in the more northern Lizard and Townsville regions. Abundances of small and large scarids were more variable over time than other taxa, but tended to decline in abundances with increasing latitudes. There also were significant regional patterns in the size and age of coral trout and red throat emperor. Both legal sized and under sized coral trout tended to be older and slightly smaller on average on southern reefs (where they were most abundant) than on northern reefs, though these patterns were attenuated on General Use reefs compared with those on MNP reefs. Red throat emperor were significantly smaller and younger in the Mackay region, where they were also most abundant, than in either the Townsville or Storm Cay regions.

Effects on Legal Sized Coral Trout

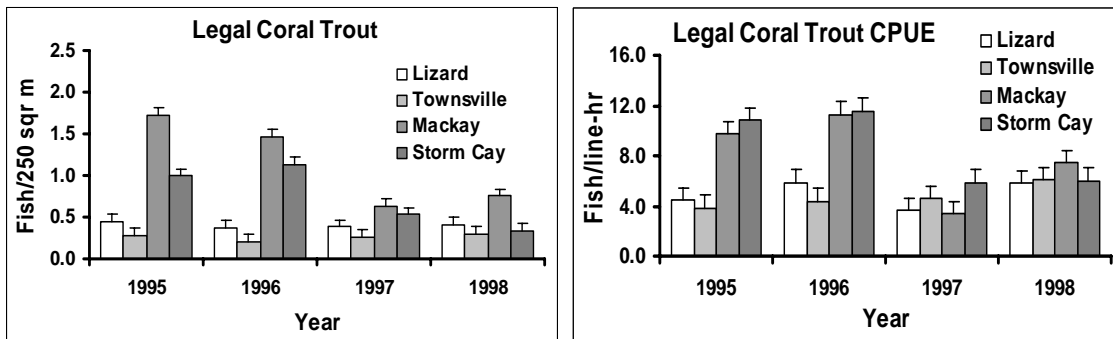
Strong regional patterns in abundance indices for legal sized coral trout were evident from both underwater visual surveys and catch surveys during baseline years. Counts of legal size coral trout were significantly and dramatically greater on reefs in the Mackay and Storm Cay regions than the Townsville and Lizard Regions in both 1995 and 1996, although the relationship between Storm Cay reefs and Mackay reefs varied between the two years (Region \times Year interaction - $F_{3,16}=4.07$, $\alpha=0.025$; Fig. 9). CPUE data showed a strong interaction between Region and Zone ($F_{3,13}=4.88$, $\alpha=0.017$) with a north-south distinction similar to the UVS data on the MNP Reefs averaged over 1995 and 1996 but no significant differences among regions on GU reefs (Fig. 9).

Figure 9: Underwater visual counts (Left) and CPUE (Right) of legal sized coral trout in each region averaged over zones in 1995 and 1996 (Counts) and averaged over years for each zone (**MNP** – Marine National Park; **GU** - General Use) (CPUE). Error bars are Standard Errors.



When surveys of control MNP and GU reefs were compared over the four years 1995-98, interactions between Region and Year were significant for both count ($F_{9,27}=6.09$, $\alpha<0.001$) and CPUE ($F_{9,36}=2.12$, $\alpha=0.053$) data. The strong regional pattern found in 1995-96, however, was absent in the later 2 years for both count and CPUE estimates of abundance (Fig. 10), there being only slightly greater estimates of density and CPUE in the two southern regions than in the northern regions (Fig. 10).

Figure 10: Underwater visual counts (Left) and CPUE (Right) of legal sized coral trout in each region averaged over non-manipulation reefs in 1995, 1996, 1997 and 1998. Error bars are Standard Errors.



Extending the comparison to 1995-2000 with data from two MNP-Control and two GU-Fished reefs in each region, there were significant interactions between Region and Year ($F_{15,40}=2.95$, $\alpha=0.003$) and Region and Zone ($F_{3,8}=7.06$, $\alpha=0.012$) for CPUE data and Region, Zone and Year for UVS data ($F_{15,45}=2.18$, $\alpha=0.023$). The strong north-south distinction was re-establishing in 1999 and 2000 in the catch rate data (Fig. 11) and the underwater count data from MNP Reefs (Fig. 12), but not in either year according to UVS data from the General Use Reefs (Fig. 12).

Figure 11: CPUE of legal sized coral trout in each region averaged over MNP Control and General Use reefs in the years 1995 – 2000 inclusive (Left) and averaged over years for each Zone (Right). Error bars are Standard Errors.

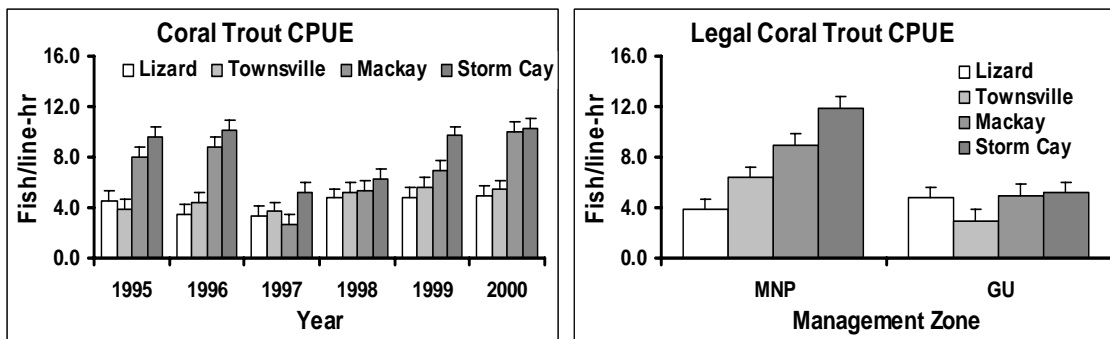
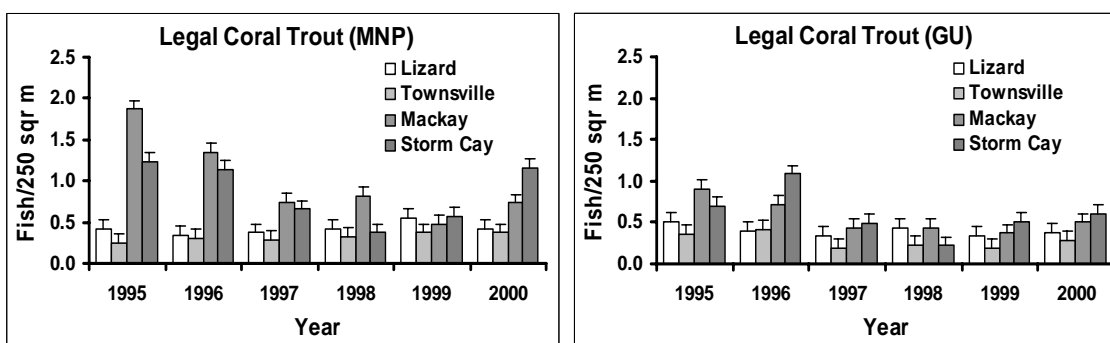


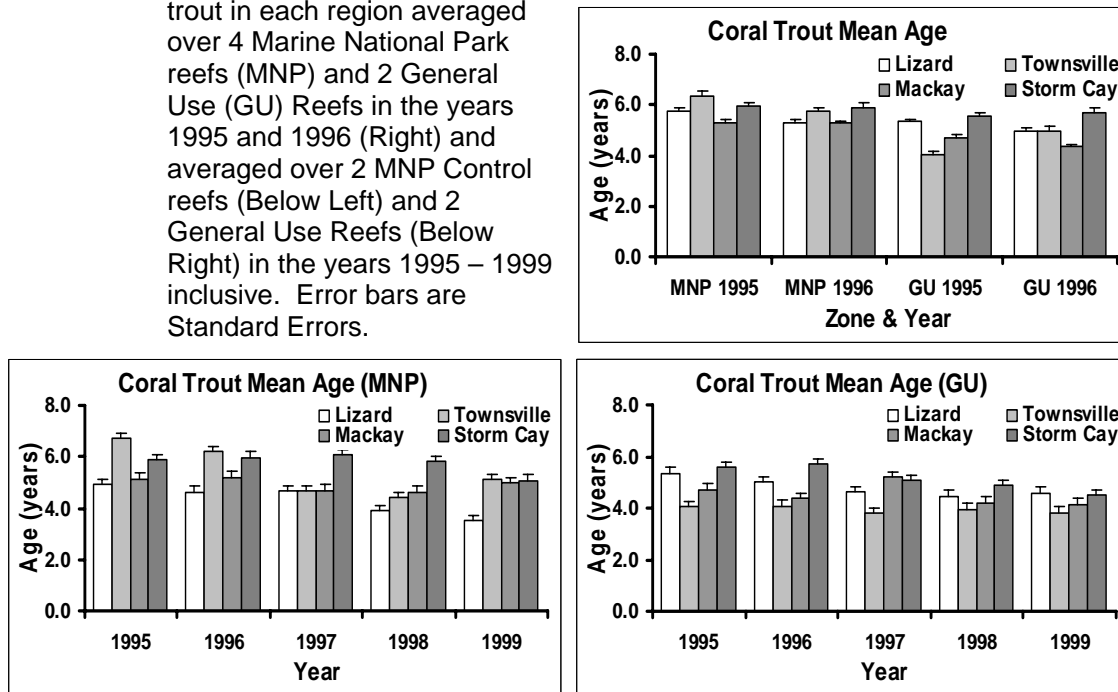
Figure 12: Underwater visual counts of legal sized coral trout in each region averaged over 2 MNP Control reefs (Left) and 2 General Use Reefs (Right) in the years 1995 – 2000 inclusive. Error bars are Standard Errors.



Interactions between Region, Zone and Year occurred during the baseline years for both mean age ($F_{3,16}=2.93$, $\alpha=0.065$) and mean length ($F_{3,12}=3.21$, $\alpha=0.062$) of legal sized coral trout and also for comparisons of age of fish from MNP-Control and GU reefs over all years (Age - $F_{12,32}=4.84$, $\alpha<0.001$).

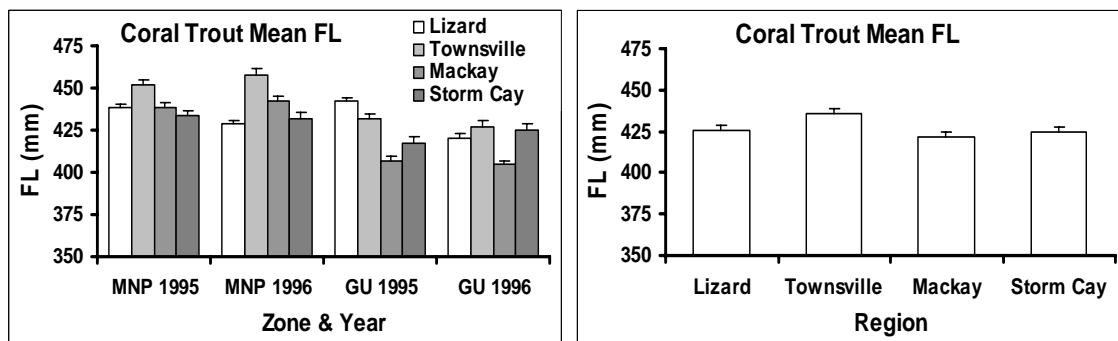
There were few strong and persistent regional patterns in mean age. Mean age of fish taken from MNP reefs was fairly consistently higher in the Storm Cay region than either the Mackay or Lizard Regions (Fig. 13), a relationship that held also for GU reefs in 1995 and 1996, but not thereafter (Fig. 13). Mean Age of legal sized coral trout on MNP Reefs in the Townsville region was highly variable, ranging from the oldest of all regions in 1995-1996 to the second lowest in 1997-98 (Fig. 13). There was a downward trend in mean age on most Lizard region reefs from 1995 to 2000 (Fig. 13).

Figure 13: Mean age of legal sized coral trout in each region averaged over 4 Marine National Park reefs (MNP) and 2 General Use (GU) Reefs in the years 1995 and 1996 (Right) and averaged over 2 MNP Control reefs (Below Left) and 2 General Use Reefs (Below Right) in the years 1995 – 1999 inclusive. Error bars are Standard Errors.



Fish from Townsville reefs tended to be significantly larger on average than fish from comparable reefs (MNP / GU) in other regions in 1995-96 and over all reefs when considered over the years 1995-2000 (Main effect of Region - $F_{3,8}=3.62, \alpha=0.065$) (Fig. 14). Harvestable fish from MNP reefs in the Lizard region were similar in size to those from the Mackay and Storm Cay regions in 1995, but significantly smaller in 1996 (Fig. 14). Averaged over all years and zones, however, fish in the Lizard region were significantly larger than those from the two southern regions (Fig. 14). Legal size coral trout on the GU reefs in the Mackay region were the smallest of all fish from GU reefs in both 1995 and 1996 (Fig. 14).

Figure 14: Mean fork length of legal sized coral trout in each region averaged over 4 Marine National Park reefs (MNP) and 2 General Use (GU) Reefs in the years 1995 and 1996 (Left) and averaged over MNP Control and GU reefs and all years (Right). Error bars are Standard Errors.

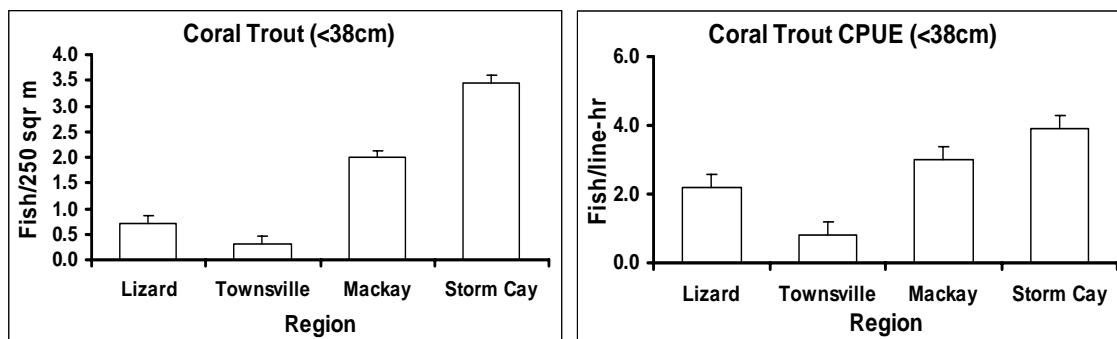


Effects on Under Sized Coral Trout

Under sized coral trout were found to be significantly and markedly (2-4 fold) more abundant in catch surveys and underwater visual surveys in the southern two regions than in the

northern regions during both baseline years (Fig. 15). This pattern was not complicated by interactions with other factors and was thus manifest as significant main effects in analyses of both CPUE ($F_{3,18}=12.96$, $\alpha<0.001$) and population density ($F_{3,19}=8.34$, $\alpha=0.009$).

Figure 15: Mean population density (Left) and CPUE (Right) of sub-legal size coral trout during baseline surveys, averaged over all reefs and 1995 and 1996 in each region. Error bars are Standard Errors.

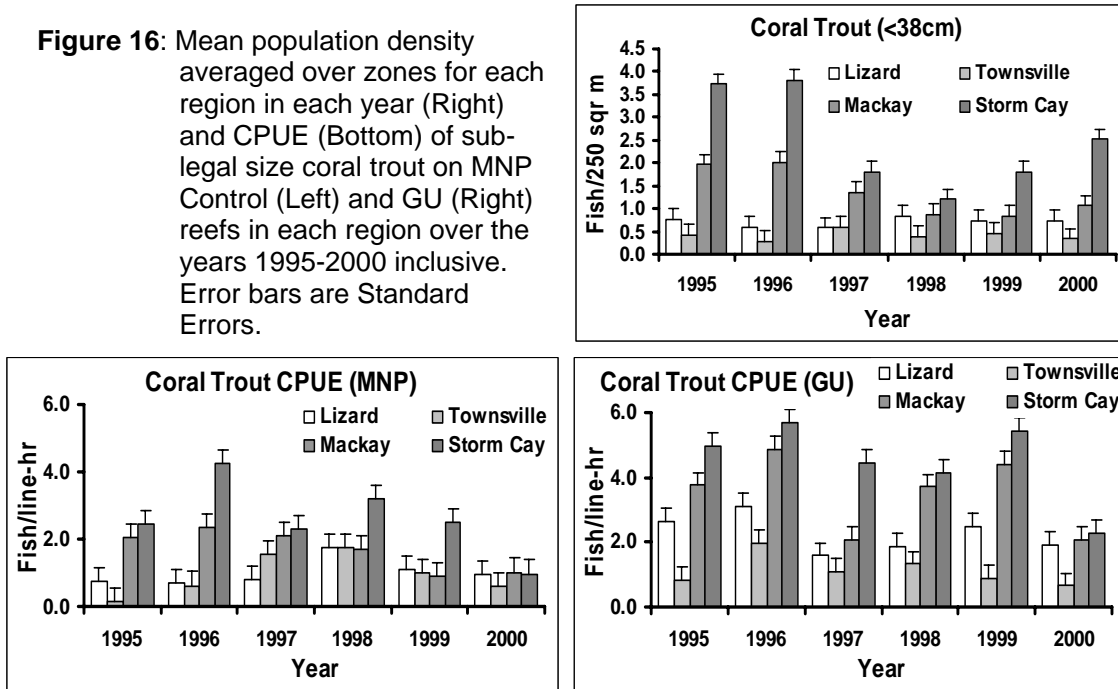


Considered over the period 1995-2000 on MNP Control and GU reefs, however, this regional contrast was not so consistent. Averaged over zones in each region and year, abundances of under sized coral trout estimated from underwater visual surveys declined sharply in the Mackay and Storm Cay Regions in 1997 and 1998 to reach levels in 1998 similar to those in the Lizard Island Region, though still greater than off Townsville (Region \times Year interaction - $F_{15,60}=5.48$, $\alpha<0.001$; Fig. 16). Abundances on Storm Cay reefs increased markedly again during 1999 and 2000, but those on Mackay reefs remained at the relatively low levels reached in 1998 and similar to levels in the Lizard region (Fig. 16). Reefs in the Townsville region consistently had the lowest population densities of sub-legal coral trout.

CPUE of under sized coral trout from catch surveys over all years also showed inter-annual variation in regional patterns, but these interactions also varied with zoning status of reefs ($F_{15,40}=1.68$, $\alpha=0.096$). The large difference in CPUE between the Storm Cay and Mackay MNP reefs on the one hand and the Lizard and Townsville MNP reefs on the other in 1995-96 diminished in 1997 because of substantial drops in CPUE in the southern two regions and a slight increase in the Townsville region (Fig. 16). During 1998 and 1999, MNP reefs around Storm Cay had significantly higher CPUE of sub-legal coral trout than on MNP reefs in the other three regions, which did not differ from each other, but in 2000 CPUE near Storm Cay also dropped again to the levels evident in other regions (Fig. 16). CPUE on Lizard and Townsville reefs in 2000 were similar to what they had been in 1995-96, after having been elevated considerably during the intervening years (Fig. 16).

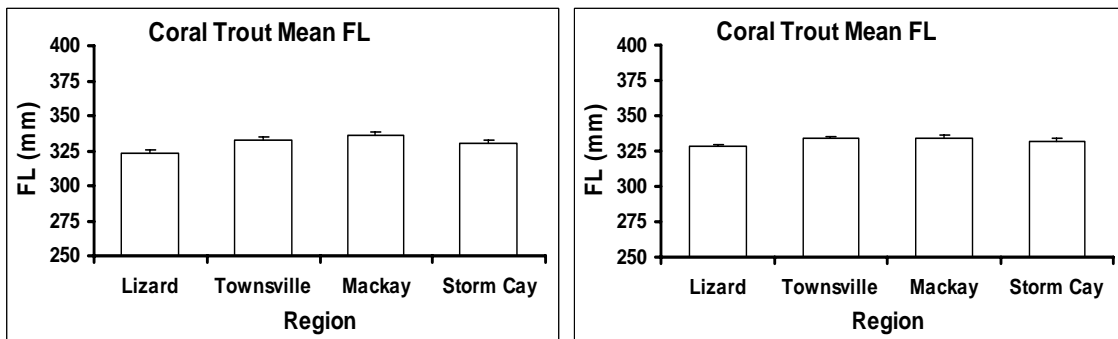
CPUE on General Use reefs in the Storm Cay region remained dramatically greater than in the northern regions from 1995 to 1999, and was similarly high on GU reefs in the Mackay Region in 1995-96 and 1998-99 (Fig. 16). CPUE in both regions plummeted in 2000 to levels similar to those on Lizard reefs, which had remained relatively stable over years (Fig. 16). CPUE of sub-legal coral trout from GU reefs off Townsville were the lowest of all regions in all years, though not significantly lower than on Lizard reefs in 1997 or 1998.

Figure 16: Mean population density averaged over zones for each region in each year (Right) and CPUE (Bottom) of sub-legal size coral trout on MNP Control (Left) and GU (Right) reefs in each region over the years 1995-2000 inclusive. Error bars are Standard Errors.



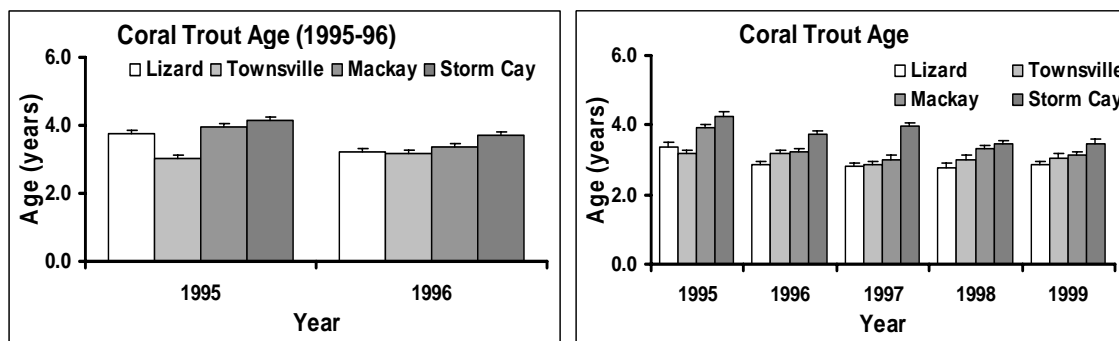
Size and age of under sized coral trout also varied with region. Mean fork length varied consistently over all reefs and years during both baseline years ($F_{3,19}=4.74$, $\alpha=0.015$) and over all years ($F_{3,8}=7.73$, $\alpha=0.094$), with fish in the Lizard region being significantly and consistently, but only slightly (~10mm, 10%), smaller than those in all other regions (Fig. 17).

Figure 17: Mean fork length of under sized coral trout in each region averaged over all reefs in 1995 and 1996 (Left) and averaged over 2 MNP Control and 2 GU reefs and all years from 1995-2000 (Right). Error bars are Standard Errors.



Age varied with Region in interaction with Year in both baseline years ($F_{3,20}=5.06$, $\alpha=0.009$) and on MNP Control and GU Reefs over all years ($F_{12,36}=2.79$, $\alpha=0.009$). The effects were consistent across zones, however, in both analyses. In 1995, under sized fish were significantly older on Mackay and Storm Cay reefs than on Lizard and Townsville reefs, and Townsville fish were significantly younger than those in all other regions (Fig. 18). By the spring of 1996, average age had decreased in the Lizard, Mackay and Storm Cay regions to near that in the Townsville region and only fish from Storm Cay were significantly older than fish elsewhere (Fig. 18). This relationship continued in the following three years (1997-99, Fig. 18).

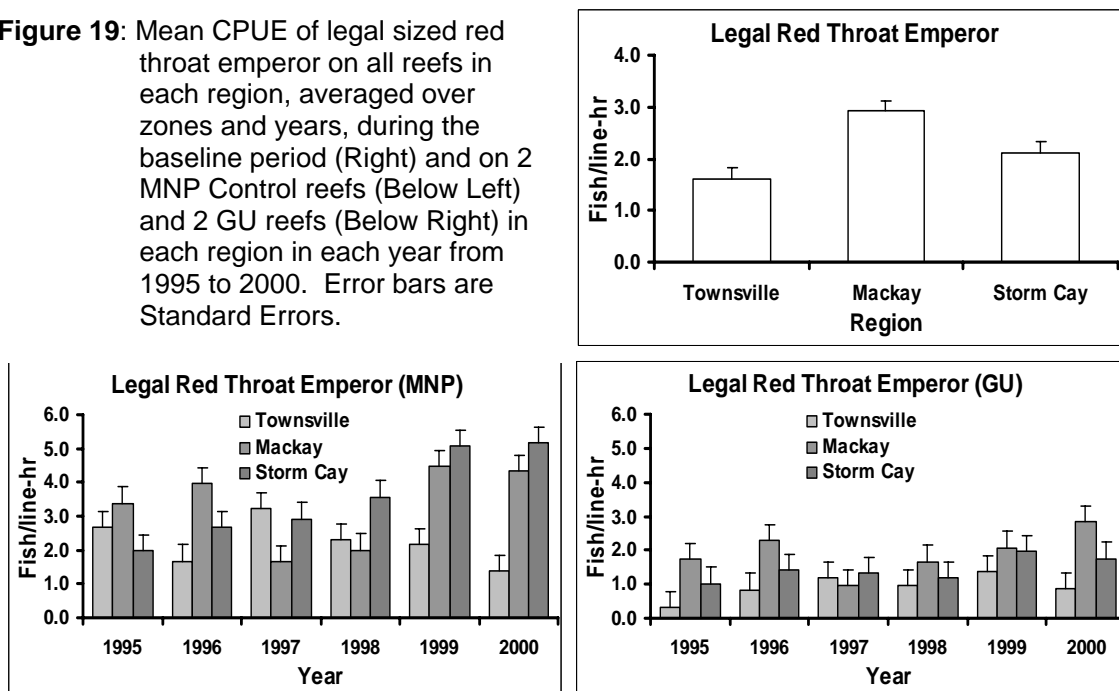
Figure 18: Mean age of under sized coral trout on all reefs in each region in 1995 and 1996 (Left) and averaged over 2 MNP Control and 2 GU Reefs in each year from 1995-1999 (Right). Error bars are Standard Errors.



Effects on Red Throat Emperor

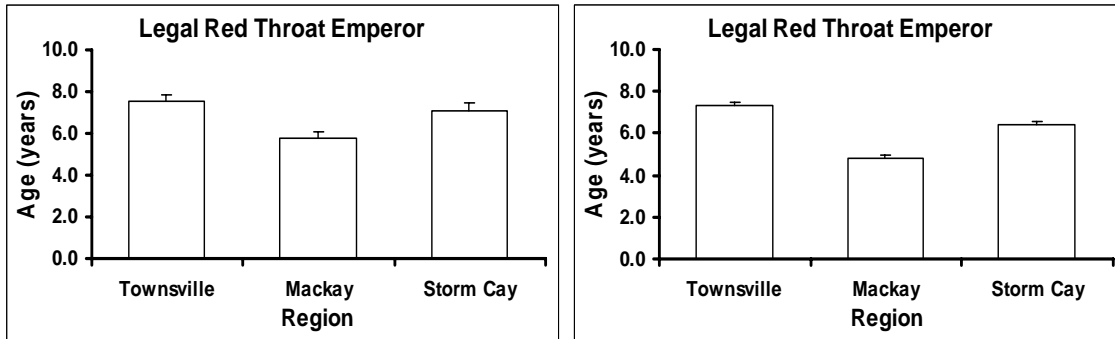
Average CPUE of harvestable red throat emperor varied significantly among regions consistently across zones and years during the baseline period (Region main effect - $F_{2,14}=10.41$, $\alpha=0.002$), but in interaction with both Year and Zone when all surveys were considered ($F_{10,35}=2.08$, $\alpha=0.054$). During the baseline surveys (1995-96), mean catch rates were significantly higher on Mackay reefs than on Storm Cay reefs, which in turn had higher catch rates than the Townsville reefs (Fig. 19). Relationships among regions varied considerably over the longer term, however. CPUE on Townsville reefs was usually the lowest of all regions, with the notable exceptions of MNP (Control) reefs in 1995, 1997 and 1998 and GU reefs in 1997 (Fig. 19). The relatively slight differences among regions in mean CPUE estimates on GU reefs mostly were not statistically significant, whilst the considerably larger differences among regions on MNP reefs usually were statistically significant (Fig. 19), with differences between the two southern regions and the Townsville region increasing after 1997 to be approximately four-fold in 2000 (Fig. 19).

Figure 19: Mean CPUE of legal sized red throat emperor on all reefs in each region, averaged over zones and years, during the baseline period (Right) and on 2 MNP Control reefs (Below Left) and 2 GU reefs (Below Right) in each region in each year from 1995 to 2000. Error bars are Standard Errors.



Regional variation in mean ages of harvestable red throat emperor were consistent across zones and years both during baseline surveys and the longer term (Baseline - $F_{2,14}=8.56$, $\alpha=0.004$; All Years - $F_{2,8}=60.65$, $\alpha<0.001$). Fish in the Mackay region were significantly younger (by 1-2 years) than in the Storm Cay and, in turn, Townsville regions (Fig. 20).

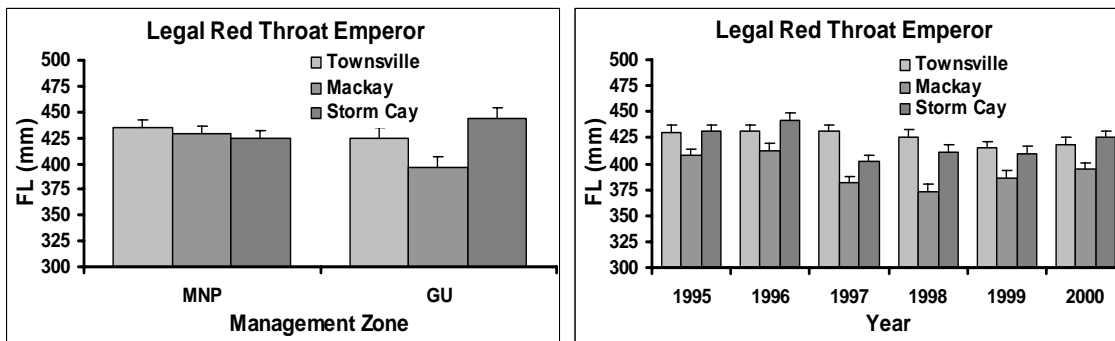
Figure 20: Mean age of legal sized red throat emperor on all reefs in each region in 1995 and 1996 (Left) and averaged over 2 MNP Control and 2 GU Reefs and all years from 1995-1999 (Right). Error bars are Standard Errors.



Average sizes of red throat emperor also varied among regions, although the nature of such differences varied with Zone during the baseline years ($F_{2,12}=4.54$, $\alpha=0.034$) and with Year over the longer term ($F_{10,45}=1.93$, $\alpha=0.066$). Sizes of fish on Mackay GU reefs were significantly smaller than on either the Townsville or Storm Cay GU reefs during the baseline surveys, but length did not differ among regions on the MNP reefs during the same period (Fig. 21). Over the longer term, regional patterns were consistent across zones, paralleling in each year the pattern seen on GU reefs in the baseline surveys (Fig. 21).

Estimated total mortality of red throat emperor on MNP reefs varied significantly among regions ($F_{2,8}=9.20$, $\alpha=0.008$), being significantly greater in the Mackay region ($Z=0.89$, $SE=0.13$) than in the Storm Cay ($Z=0.61$, $SE=0.11$) and Townsville ($Z=0.31$, $SE=0.10$) regions, which also differed from each other.

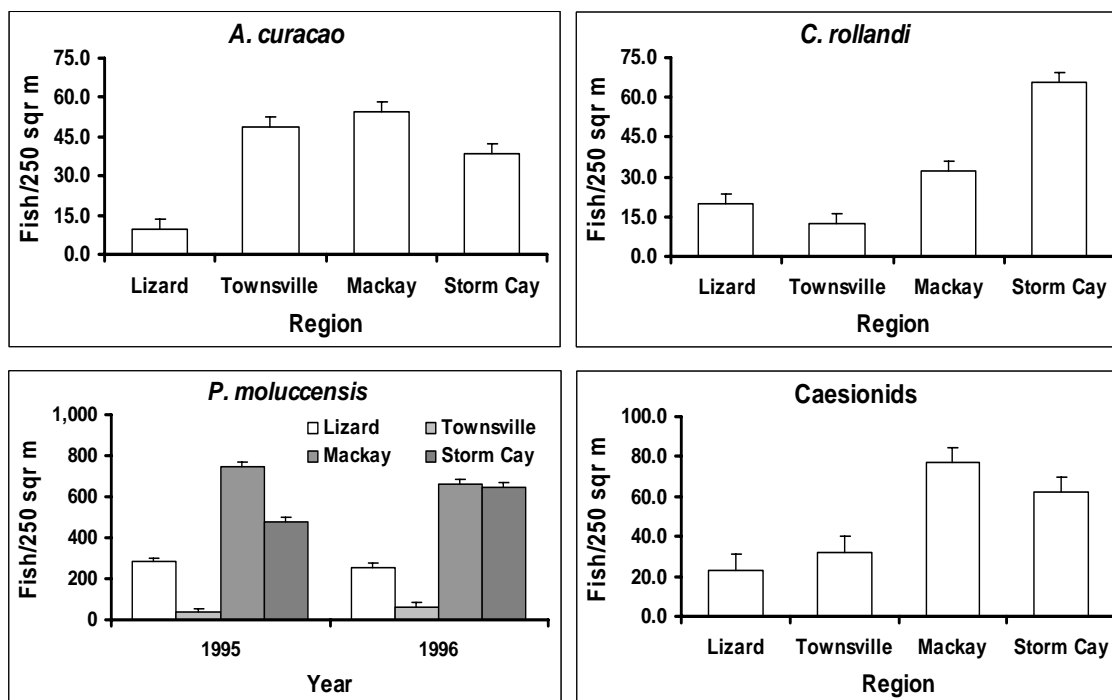
Figure 21: Mean fork length (FL) of legal sized red throat emperor in each region on 4 MNP reefs and 2 GU reefs averaged over 1995 and 1996 (Left) and averaged over 2 MNP Control and 2 GU Reefs in each year from 1995-1999 (Right). Error bars are Standard Errors.



Effects on Non-catch Species

There were significant regional patterns in abundances of non-catch species both during baseline surveys of all reefs (1995-96) and surveys of two MNP Control and 2 General Use reefs in each cluster from 1995-2000.

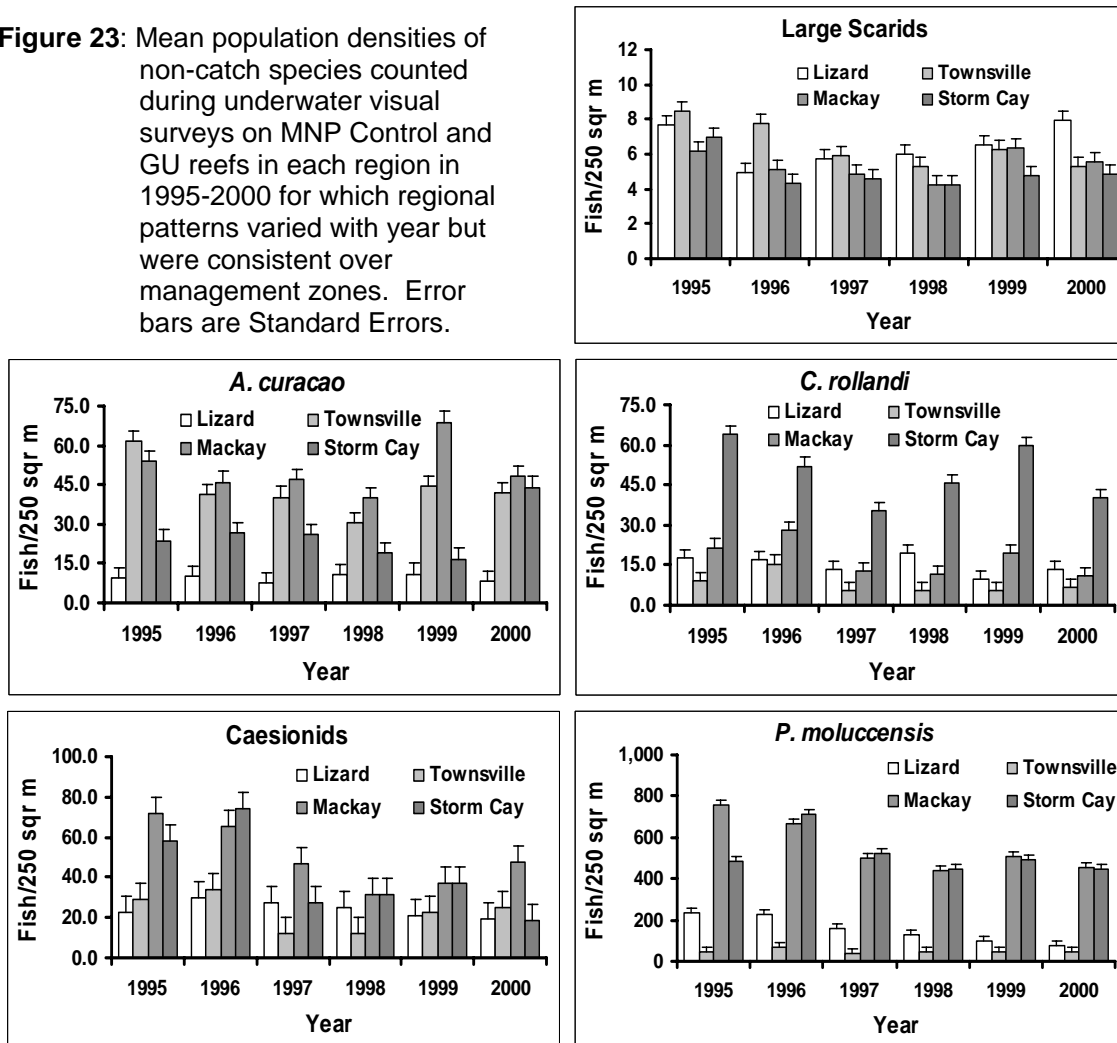
Figure 22: Mean population densities of non-catch species counted during underwater visual surveys on all reefs in each region in 1995 and 1996 for which patterns were consistent over management zones. Error bars are Standard Errors.



Main effects of region were significant and uncomplicated during baseline years for *A. curacao* ($F_{3,19}=24.33$, $\alpha<0.001$), *C. rollandi* ($F_{3,18}=32.34$, $\alpha<0.001$) and caesionids ($F_{3,19}=10.27$, $\alpha<0.001$), whilst there was a significant Region x Year interaction for *P. moluccensis* ($F_{3,15}=15.79$, $\alpha<0.001$). All these taxa were significantly less abundant in the Lizard region than in the Mackay and Storm Cay regions, and *C. rollandi*, *P. moluccensis* and caesionids had abundances in the Townsville region similar to or lower than those in the Lizard region (Fig. 22). Abundances of *A. curacao* were similar on Townsville and Mackay reefs and significantly greater in all three southernmost regions than on reefs in the Lizard region (Fig. 22).

Most of these patterns were consistent over subsequent years, despite significant Region x Year interactions for all four groups. *A. curacao* ($F_{15,60}=3.12$, $\alpha=0.001$) was consistently at lowest abundances in the Lizard region, greatest abundances in the Townsville and Mackay regions and (usually) intermediate abundances around Storm Cay. There was consistently a 3-5 fold or greater difference between counts from Lizard reefs and those from Townsville and Mackay reefs (Fig. 23). Storm Cay reefs consistently had 2-4 times as many *C. rollandi* as reefs in the other regions, amongst which relative densities varied over time ($F_{15,55}=3.27$, $\alpha=0.001$; Fig. 23). *P. moluccensis* ($F_{15,60}=8.50$, $\alpha<0.001$) was consistently at least 4 times as abundant in the Mackay and Storm Cay regions as in the Lizard and Townsville region (Fig. 23). Caesionids showed a significant interaction between Year and Region ($F_{15,60}=1.64$, $\alpha=0.090$), with significantly greater abundances in the southern regions in 1995 and 1996 but few significant differences among regions in most years thereafter (Fig. 23).

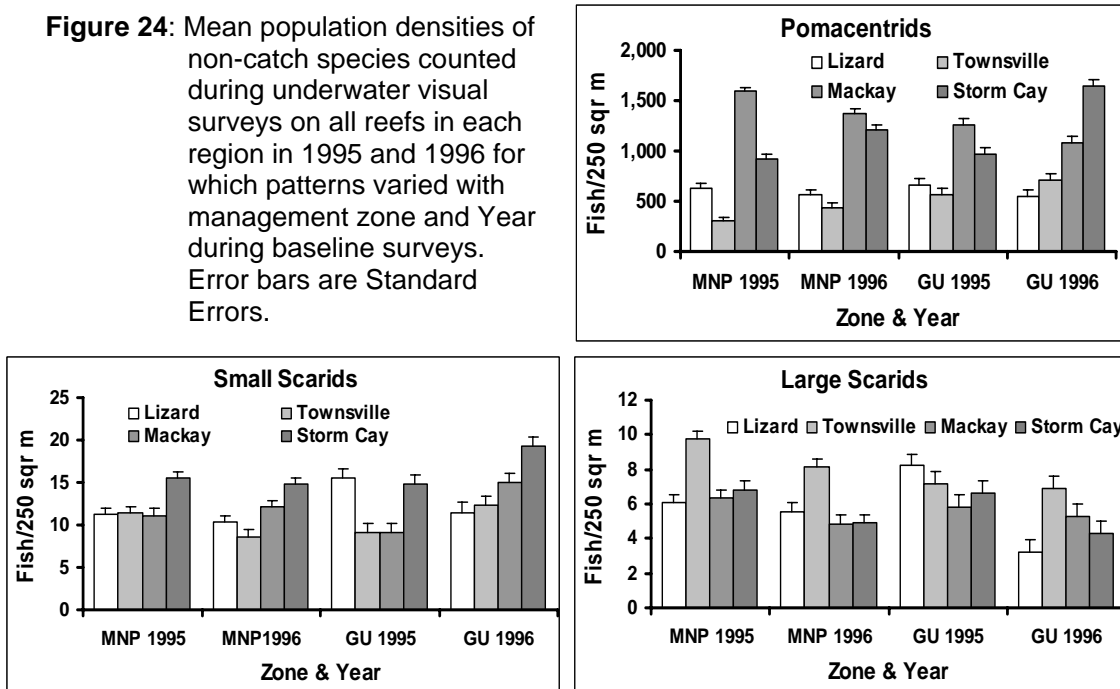
Figure 23: Mean population densities of non-catch species counted during underwater visual surveys on MNP Control and GU reefs in each region in 1995-2000 for which regional patterns varied with year but were consistent over management zones. Error bars are Standard Errors.



Large Scarids also varied significantly with Region and Year ($F_{15,55}=2.32$, $\alpha=0.012$) and although generally more regionally uniform in abundance than the above taxa, tended to be least abundant around Storm Cay and most abundant in Lizard and / or Townsville regions in most years (Fig. 23).

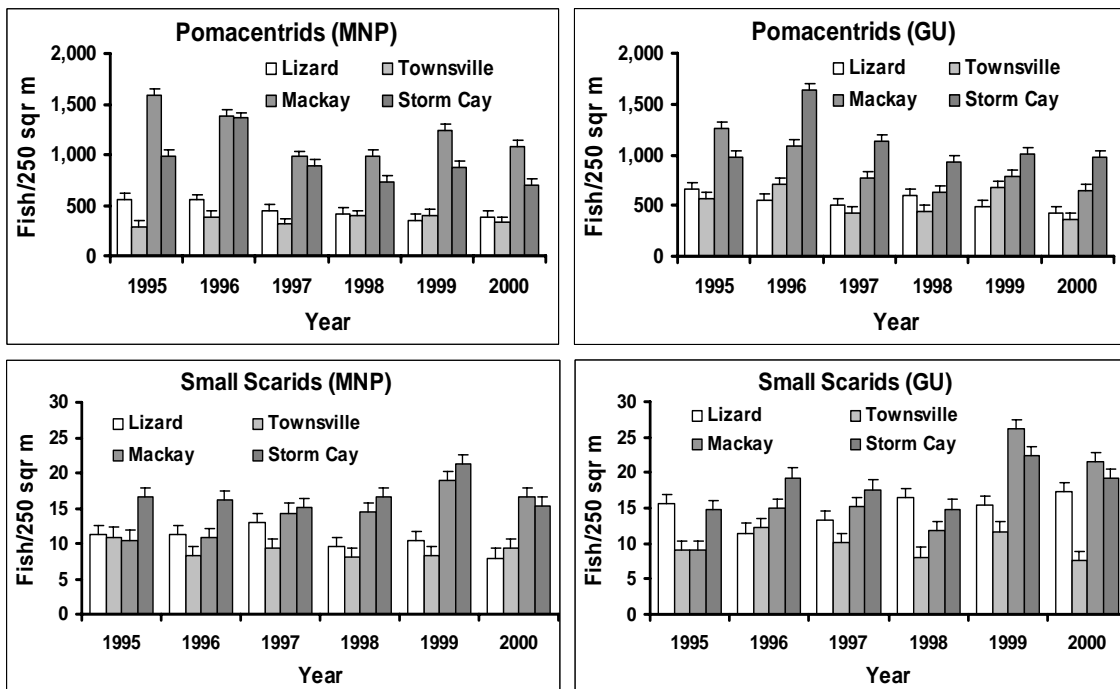
Interactions among Region, Zone and Year significantly affected abundances of large scarids during baseline surveys ($F_{3,20}=4.64$, $\alpha=0.013$) and abundances of mixed pomacentrids and small scarids during the baseline period (Pomacentrids - $F_{3,16}=3.077$, $\alpha=0.058$; small scarids - $F_{3,16}=5.27$, $\alpha=0.010$) and in the longer term (Pomacentrids - $F_{15,45}=1.71$, $\alpha=0.083$; small scarids - $F_{15,40}=2.58$, $\alpha=0.009$). Despite these interactions, the pomacentrids were consistently more abundant in the southern two regions than in the northern two regions (Fig. 24, Fig. 25). The differences were larger on MNP reefs than GU reefs, but were significant in all cases except for the contrast between Mackay region reefs and Townsville reefs in 1999 and Lizard reefs in 1998 (Fig. 24, Fig. 25). The relationships between Lizard and Townsville reefs varied with time in both zones, as did the relationship between Mackay and Storm Cay reefs (Fig. 25).

Figure 24: Mean population densities of non-catch species counted during underwater visual surveys on all reefs in each region in 1995 and 1996 for which patterns varied with management zone and Year during baseline surveys. Error bars are Standard Errors.



Although small scarids often were more abundant in southern regions than in northern regions (Fig. 24, Fig. 25), regional patterns in abundances of both small and large scarids were not consistent over years, either in 1996-96 (Fig. 24) or the longer term (Fig. 25).

Figure 25: Mean population densities of non-catch species from underwater visual surveys on MNP reefs and GU reefs in each region in 1995-2000 for which regional patterns varied with Zone and Year. Error bars are Standard Errors.



Effects of Management Zone

Effects of prior zoning status (open or closed to fishing) were evident for abundances, size and age of both coral trout and red throat emperor, but the effects varied regionally and, in some cases, over time. Legal sized coral trout were fairly consistently more abundant, larger and slightly older on MNP (closed) reefs than on GU (open) reefs in the Townsville, Mackay and Storm Cay regions, but usually similar in most respects on open and closed reefs around Lizard Island. In general, the magnitude of the differences between zones increased with latitude. Similarly, legal sized red throat emperor were more abundant on the closed reefs, though the effects of zoning on age and size of red throat emperor were variable. Conversely, under sized coral trout were most often significantly more abundant on the GU reefs than the MNP reefs. Relationships between zoning history and abundances of non-catch species were not consistent over years or regions.

Effects on Legal Sized Coral Trout

Underwater counts of legal sized coral trout during baseline surveys (1995-96) differed between reefs zoned Marine National Park and reefs zoned General Use, but the differences were region specific (Region x Zone interaction - $F_{3,13}=10.37, \alpha=0.001$). Average densities of legal size coral trout on the four MNP reefs were significantly greater (by more than 100%) than on the two General Use Reefs in the Mackay Region, but did not differ significantly with zoning history in the other regions (Fig. 26). Results were slightly more complex when the two MNP Control reefs were compared to the two GU reefs in each region over the six years 1995-2000 (Region x Year x Zone interaction - $F_{15,45}=2.18, \alpha=0.023$). Counts were significantly greater on MNP reefs than on GU reefs in the Lizard Region only in 1999, not at all in the Townsville region, in the Mackay region in all years except 1999 and in the Storm Cay Region in 1995 and 2000 (Fig. 27).

Figure 26: Underwater visual counts of legal sized coral trout (>38cm TL) averaged over 1995 and 1996 from reefs zoned Marine National Park (MNP) and General Use (GU) in each region. Error bars are Standard Errors.

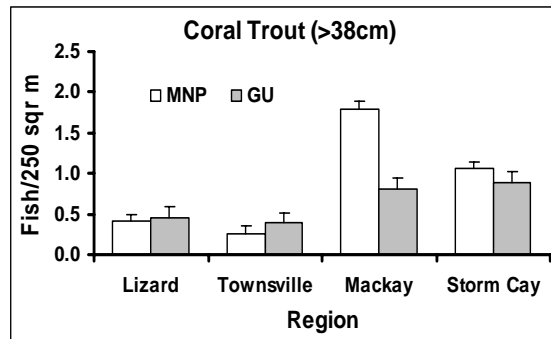
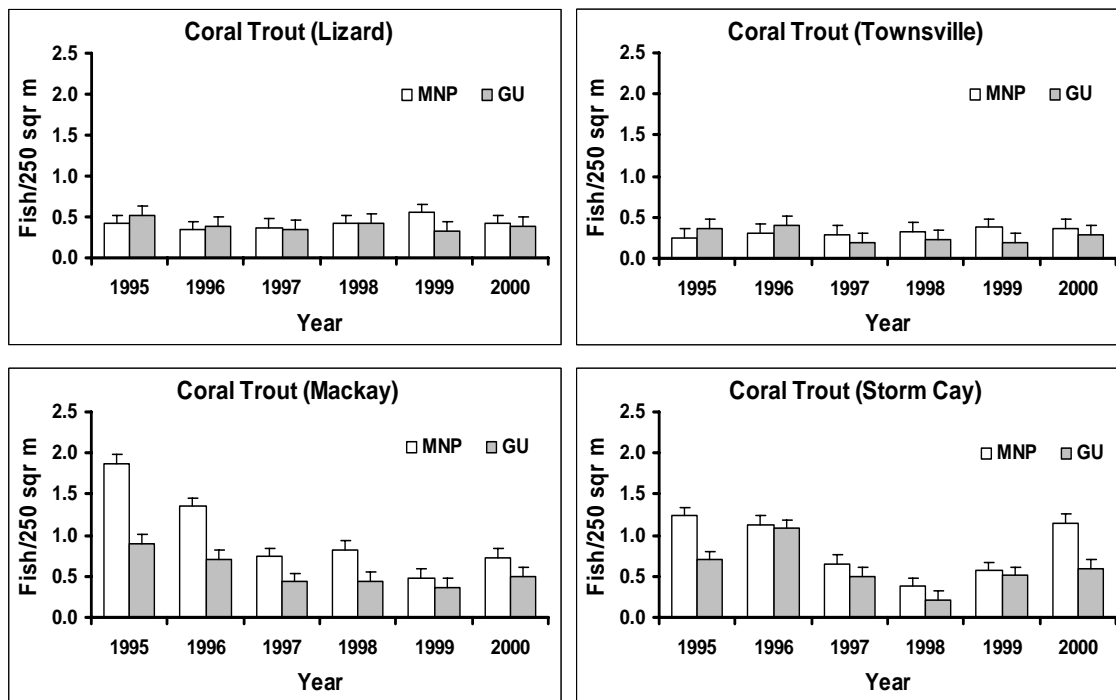
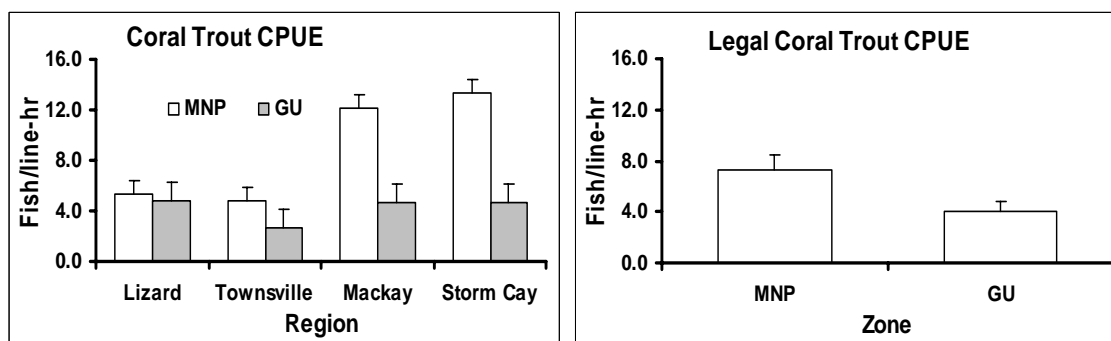


Figure 27: Underwater visual counts of legal sized coral trout (>38cm TL) in 1995-2000 from 2 reefs zoned Marine National Park (MNP) and 2 reefs zoned General Use (GU) in each region. Error bars are Standard Errors.



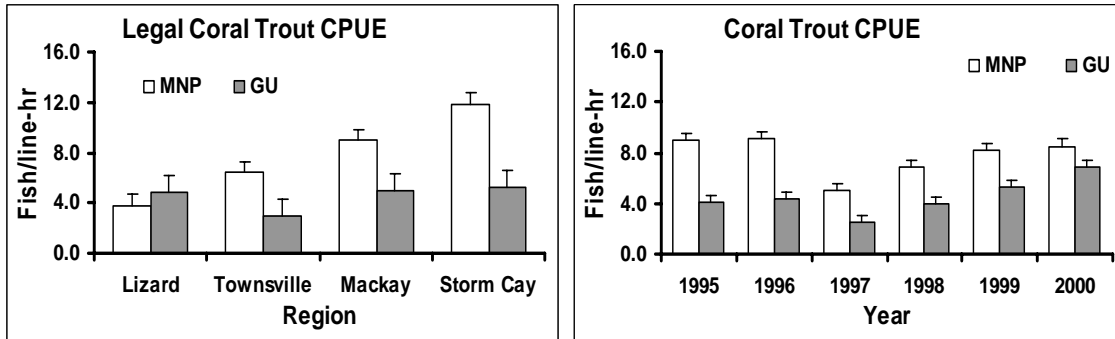
Catch rates of legal sized coral trout also varied significantly between Marine National Park and General Use reefs during baseline surveys, though the differences varied with region (Zone x Region interaction - $F_{3,13}=4.88$, $\alpha=0.017$). During 1995-96, CPUE of legal coral trout was almost identical on MNP and GU reefs in the Lizard Region, was greater, though not significantly so, on MNP reefs than on GU reefs off Townsville and significantly and dramatically higher (by about 3-fold) on MNP reefs than on GU reefs in the Mackay and Storm Cay regions (Fig. 28). When three MNP reefs were compared with a single GU reef in each region over the years 1995-98, the difference between the MNP and GU reefs was consistent among regions and over years (Main effect of Zone - $F_{1,11}=5.92$, $\alpha=0.033$), with the overall average CPUE being around twice as high on the closed reefs (Fig. 28).

Figure 28: Mean CPUE of legal sized coral trout (>38cm TL) from reefs zoned Marine National Park (MNP) and General Use (GU) in each region averaged over 1995 and 1996 (Left) and averaged over 1995 – 1998 and all regions (Right). Error bars are Standard Errors.



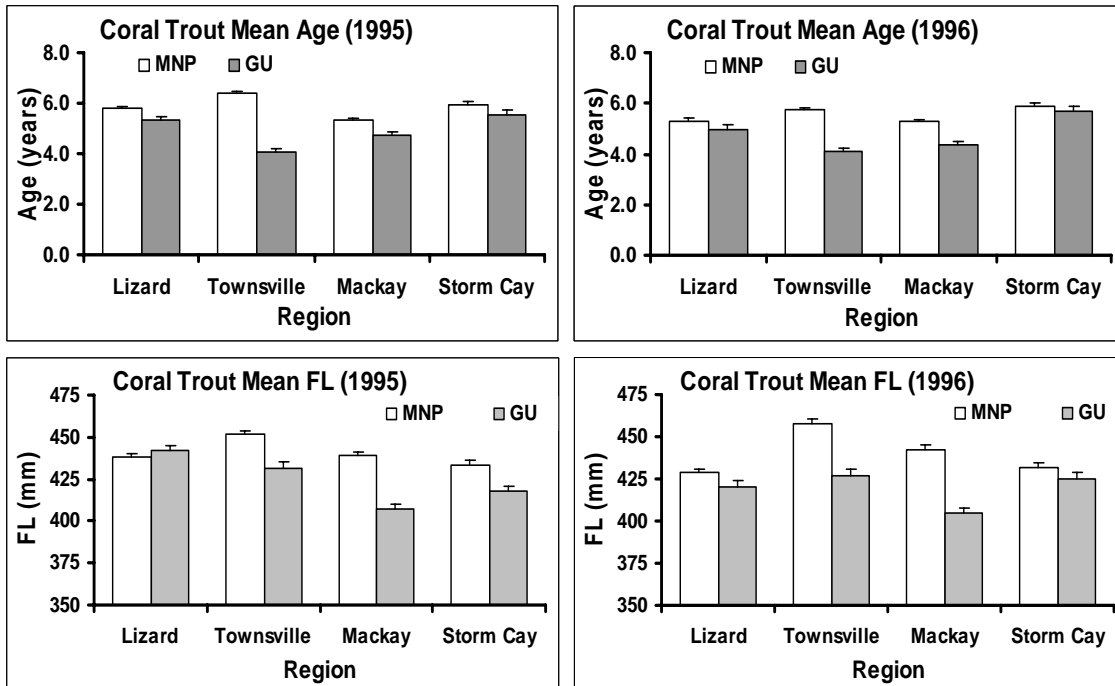
Considered over all years 1995-2000, catch rates of legal sized coral trout varied in interactions between Zone and Region ($F_{3,8}=7.06$, $\alpha=0.012$) and Zone and Year ($F_{5,40}=2.75$, $\alpha=0.032$). Averaged over all years, CPUE on the two MNP Control reefs was approximately 2-2.5 times as great as on the two GU reefs in the Townsville, Mackay and Storm Cay Regions, but did not differ between zones in the Lizard region (Fig. 29). Averaged over all regions, catch rates were significantly greater on MNP reefs than on GU reefs in all years, with the differences being at least two-fold in 1995-97, and then diminishing to less than 25% in 2000 (Fig. 29).

Figure 29: Mean CPUE for legal sized coral trout from 2 unfished Marine National Park (MNP) and 2 General Use (GU) reefs in each region sampled between 1995 and 2000. Left - Means for the Zone x Region interaction; Right – Means for the Zone x Year interaction. Error bars are Standard Errors.



Average age and size of harvestable coral trout also varied with zoning status. Fish on GU reefs were never significantly older or larger than those on MNP reefs, but both the magnitude and significance of differences between zones varied with regions and years.

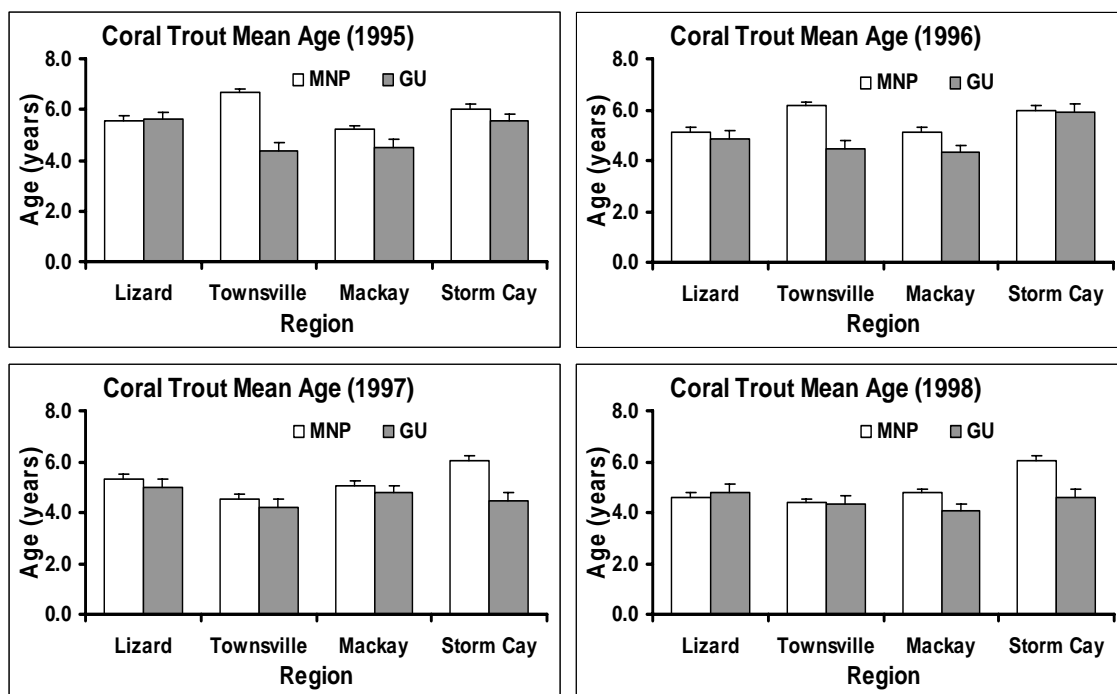
Figure 30: Mean age (Top) and length (Bottom) of legal sized coral trout (>38cm TL) from reefs zoned Marine National Park (MNP) and General Use (GU) in each region in 1995 (left) and 1996 (right). Error bars are Standard Errors.



During the baseline years of the experiment, significant interactions between Year, Region and Zone were evident for both mean age ($F_{3,16}=2.93$, $\alpha=0.065$) and mean FL ($F_{3,12}=3.21$, $\alpha=0.062$). Legal sized coral trout were on average significantly older and larger on MNP reefs than on GU reefs in both the Townsville and Mackay Regions in both 1995 and 1996 (Fig. 30) and larger but not older on MNP reefs in the Storm Cay region in both years (Fig. 30). Coral trout were significantly older but not significantly larger on MNP reefs in the Lizard Region in 1995 and larger but not older in 1996 (Fig. 30). The magnitudes of differences in age ranged from nearly two years (50%) in the Townsville region to almost nothing in the Storm Cay region, whilst differences in length ranged from about 40mm (10%) in the Mackay Region to just a few millimetres in the Lizard Region (Fig. 30).

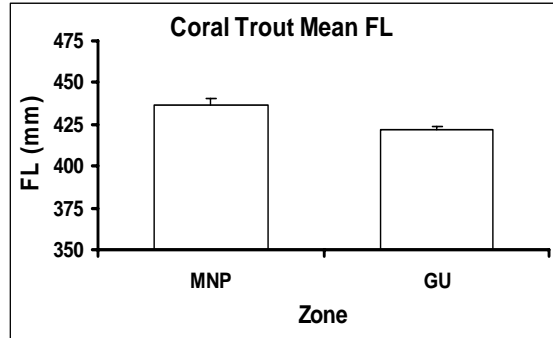
A significant interaction between Year, Zone and Region resulted when ages of legal sized coral trout from un-fished MNP and GU reefs were compared over the four years 1995-98 ($F_{9,27}=4.51$, $\alpha=0.001$). Again, mean ages on GU reefs never significantly exceeded those on MNP reefs, but ages on MNP reefs significantly exceeded those on GU reefs only in the Townsville and Mackay Region in 1995 and 1996 and in the Storm Cay Region in 1997 and 1998 (Fig. 31).

Figure 31: Mean age of legal sized coral trout (>38cm TL) from unfished reefs zoned Marine National Park (MNP) and General Use (GU) reefs in each region in 1995, 1996, 1997 and 1998. Error bars are Standard Errors.



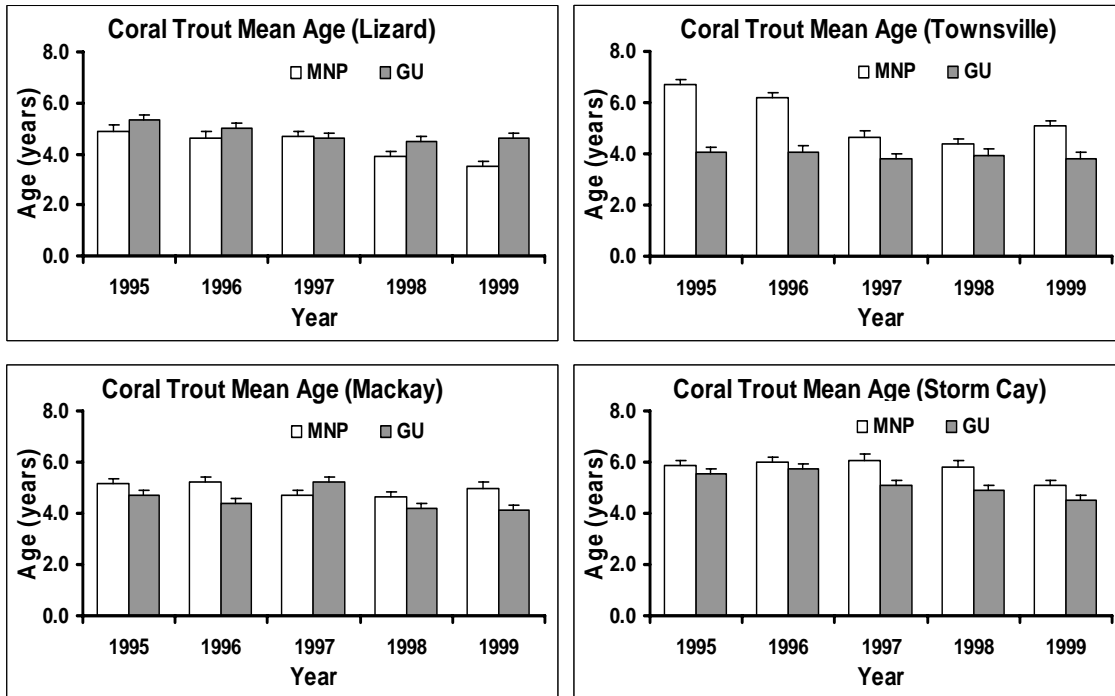
Fork lengths of legal sized coral trout from the same reefs and years differed with zone consistently over years and regions (main effect of Zone - $F_{1,8}=11.01$, $\alpha=0.011$), with fish being approximately 20mm (5%) longer on the closed reefs than on the reefs open to fishing (Fig. 32).

Figure 32: Mean length of legal sized coral trout (>38cm TL) from unfished reefs zoned Marine National Park (MNP) and General Use (GU) reefs averaged over all regions and the years 1995-98. Error bars are Standard Errors.



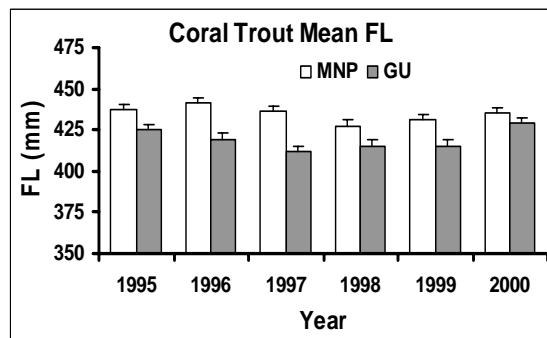
An interaction among Year, Zone and Region also was significant when mean ages of legal sized coral trout were compared on the MNP Control and GU reefs over the years 1995-99 ($F_{12,32}=4.84$, $\alpha<0.001$). Harvestable coral trout were significantly older on the GU reefs than on the MNP Control reefs in 1998 and 1999 in the Lizard Region, but similar in both zones in the Lizard region in the other years (Fig. 33). Fish were significantly older on average on MNP Control reefs than on GU reefs off Townsville in all years except 1998, in the Mackay Region in 1995, 1996 and 1999, and in the Storm Cay Region in 1997-1999 (Fig. 33). Apart from in 1995 and 1996 off Townsville, the differences in mean ages were small (<1year).

Figure 33: Mean age of legal sized coral trout (>38cm TL) from 2 unfished reefs zoned Marine National Park (MNP) and 2 General Use (GU) reefs in each region in 1995, 1996, 1997, 1998 and 1999. Error bars are Standard Errors.



The pattern in average length of coral trout across management zones varied with year but was consistent over regions over the years 1995-2000 ($F_{5,70}=1.98$, $\alpha=0.093$). Fish were on average larger by about 20mm on MNP Control Reefs than on GU Reefs in all years up to and including 1999, but did not differ between zones in 2000 (Fig. 34).

Figure 34: Mean length of legal sized coral trout (>38cm TL) from unfished reefs zoned Marine National Park (MNP) and General Use (GU) reefs averaged over all regions in the years 1995-99. Error bars are Standard Errors.



Estimates of total mortality of coral trout considered fully recruited to the gear (those aged 4 years and older) were lower on MNP reefs than GU reefs in each region and were generally greatest in the Storm Cay region and least in the Townsville region in both zones (Table 4). None of these apparent differences were statistically significant, however ($P > 0.1$ in all tests).

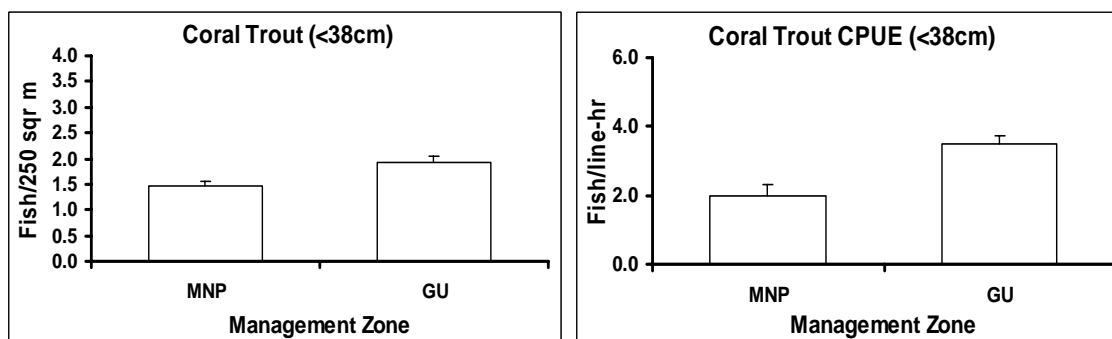
Table 4: Estimated total mortality (Z) of fully recruited coral trout on un-manipulated MNP reefs and GU reefs in each region over the years 1995-99.

Region	Zone			
	MNP		GU	
	Mortality	(SE)	Mortality	(SE)
Lizard	0.61	(0.14)	0.63	(0.14)
Townsville	0.45	(0.11)	0.58	(0.23)
Mackay	0.61	(0.11)	0.75	(0.15)
Storm Cay	0.84	(0.11)	0.93	(0.18)

Effects on Under Sized Coral Trout

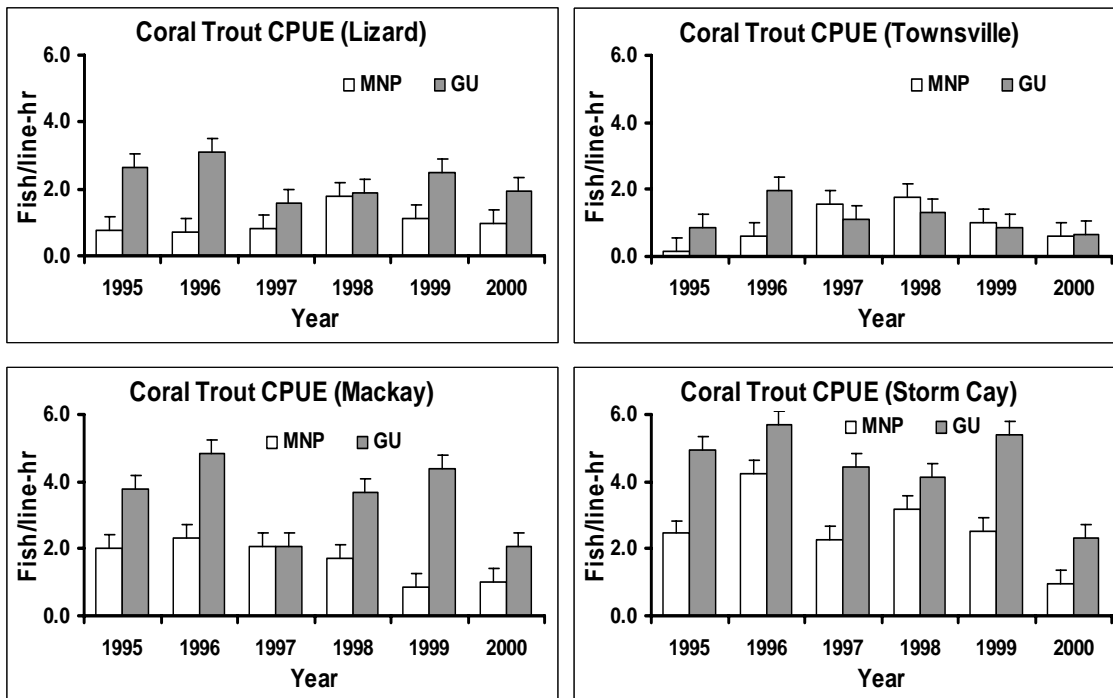
Both underwater visual counts and catch surveys indicated significantly lower abundances of undersize coral trout on MNP reefs than on GU reefs during 1995 and 1996. Moreover, both results were consistent over both regions and years, leaving clear main effects of zones (Visual Surveys - $F_{1,19}=8.34$, $\alpha=0.009$; Catch Surveys - $F_{1,18}=14.51$, $\alpha=0.001$) (Fig. 35). The same main-effect result was evident in the underwater visual survey data for MNP Control and GU reefs over the years 1995-2000 ($F_{1,11}=6.97$, $\alpha=0.023$; GU reefs - 1.32 fish/250m²; MNP reefs - 1.15 fish/250m²).

Figure 35: Mean density estimated from underwater visual surveys (Left) and CPUE from structured catch surveys (Right) of sub-legal coral trout (<38cm TL) from reefs zoned Marine National Park (MNP) and General Use (GU) averaged over all regions and 1995-1996. Error bars are Standard Errors.



CPUE of sub-legal coral trout from MNP Control and GU reefs over the years 1995-2000 varied with Region, Zone and Year ($F_{15,40}=1.68$, $\alpha=0.096$). For the most part, the same pattern was evident as during the baseline years, with CPUE of under sized coral trout never being significantly greater on MNP Control reefs than on GU Reefs. Catch rates were significantly greater on GU reefs than on unfished MNP reefs in Lizard, Mackay and Storm Cay Region in all years except 1997 and 1998 (Lizard Region), 1997 (Mackay) and 1998 (Storm Cay) (Fig. 36). In the Townsville Region, catch rates were significantly greater on the GU reefs in 1995 and 1996, but no other years (Fig. 36).

Figure 36: Mean CPUE of sub-legal coral trout (<38cm TL) from Marine National Park (MNP) Control and General Use (GU) reefs in each region in each year from 1995-2000. Error bars are Standard Errors.

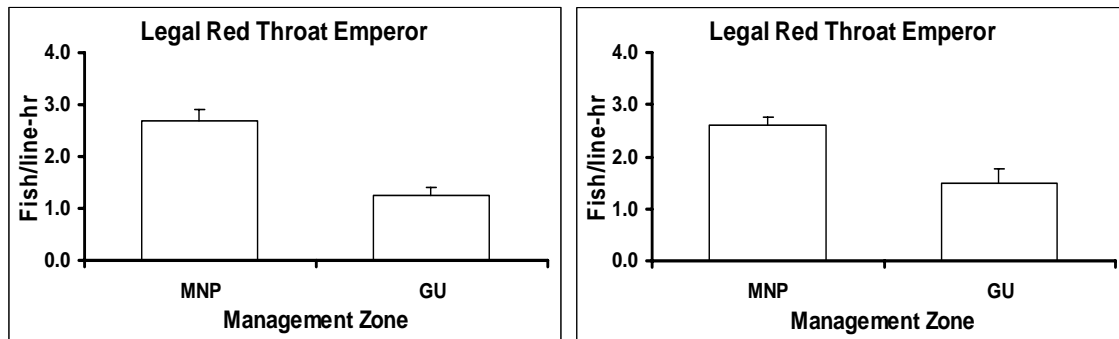


There were no significant effects of zoning on mean age or fork length of sub-legal coral trout ($P>0.1$ in all tests).

Effects on Red Throat Emperor

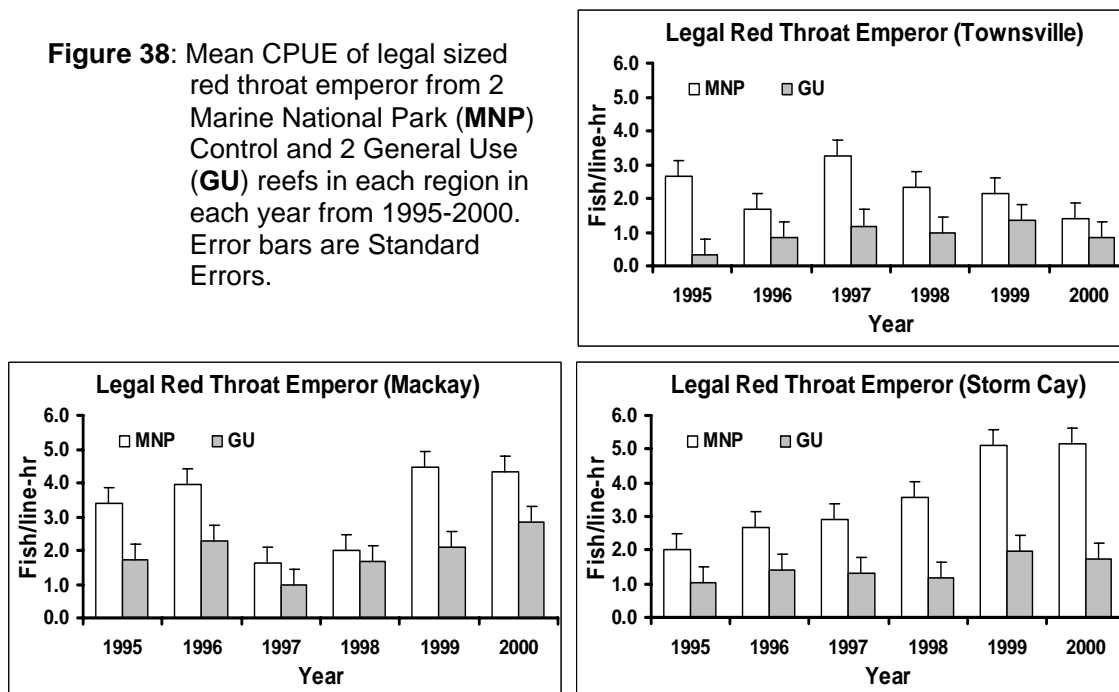
There were conspicuous and relatively uncomplicated differences in CPUE of red throat emperor between reefs zoned Marine National Park (MNP) and those zoned General Use (GU) in all regions. During baseline surveys, effects of management zone were consistent over years and regions (Zone main effect - $F_{1,14}=33.71$, $\alpha<0.001$), overall average CPUE being nearly three times as high on 16 MNP reefs as on the 8 GU reefs (Fig. 37). A parallel result arose from comparing 3 MNP control and 2 GU reefs from each region over the years 1995-98 ($F_{1,8}=11.18$, $\alpha=0.010$; Fig 37).

Figure 37: Mean CPUE of legal sized red throat emperor from un-manipulated reefs zoned Marine National Park (**MNP**) and General Use (**GU**) averaged over 1995-1996 (Left) and 1995 – 1998 (Right) and all regions. Error bars are Standard Errors.



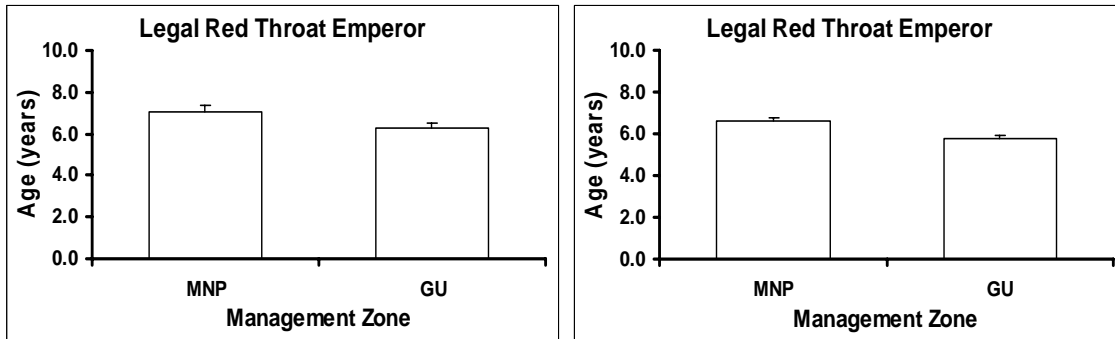
Comparing CPUE of red throat emperor on 2 MNP Control and 2 GU reefs from each region sampled from 1995-2000 (inclusive) resulted in a significant interaction between management Zone, Region and Year ($F_{10,35}=2.08$, $\alpha=0.054$). Whilst CPUE from unfished Marine National Park reefs was consistently greater than on nearby GU reefs (Fig. 38), the effects were not statistically significant in all years in all regions. CPUE was statistically greater on MNP reefs than on GU reefs in the Townsville region in 1995, 1997 and 1998, in the Mackay region in 1995, 1996, 1999 and 2000 and in the Storm Cay region in all years (Fig. 38), with the differences in the Storm Cay region increasing with time. The magnitude of (significant) differences between zones ranged from approximately 2-fold to nearly 5-fold.

Figure 38: Mean CPUE of legal sized red throat emperor from 2 Marine National Park (**MNP**) Control and 2 General Use (**GU**) reefs in each region in each year from 1995-2000. Error bars are Standard Errors.



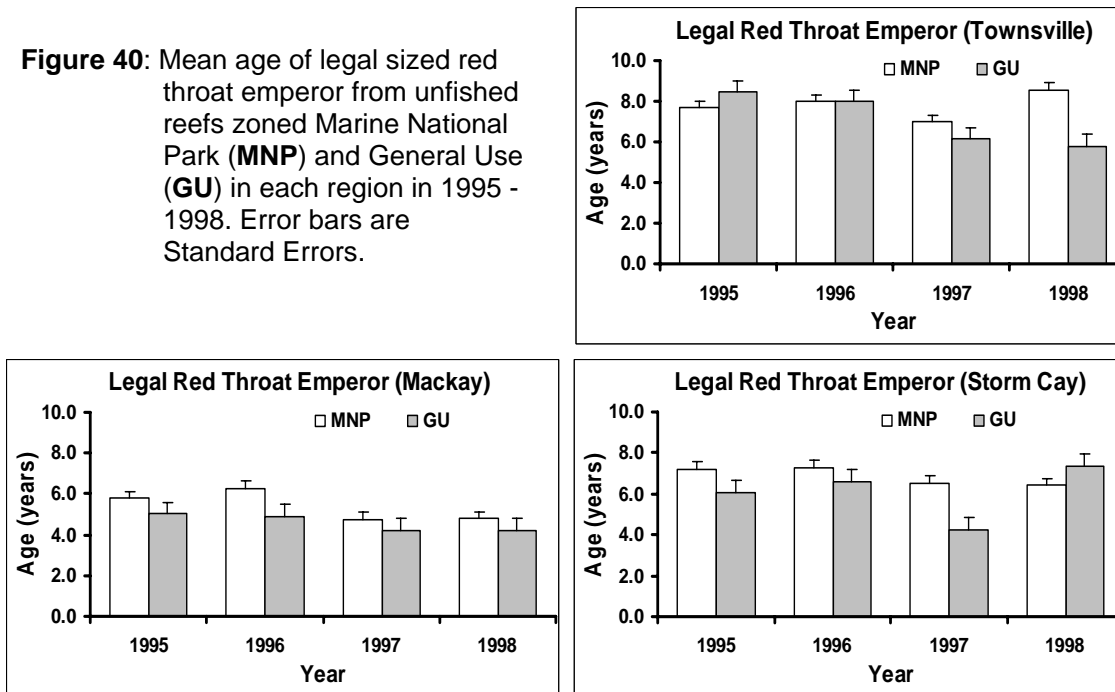
Mean age and size of harvestable red throat emperor also varied significantly with past zoning status. Fish were significantly older on average by approximately one year on MNP reefs than on General Use reefs averaged over all regions and years during baseline surveys (Zone main effect - $F_{1,14}=4.21$, $\alpha=0.059$) and all surveys between 1995 and 2000 (Zone main effect - $F_{1,8}=20.99$, $\alpha=0.002$) (Fig. 39).

Figure 39: Mean age of legal sized red throat emperor from Marine National Park (MNP) and General Use (GU) reefs averaged over all regions and years from surveys in 1995-1996 (Left) and 1995-2000 (Right). Error bars are Standard Errors.



Contrasts between MNP and single GU reefs in each region between 1995 and 1998 were less consistent, age of red throat emperor varying significantly among years and regions ($F_{6,27}=4.02, \alpha=0.005$). Mostly, however, patterns accorded with those identified from baseline and longer-term surveys (Fig. 40). Fish from MNP reefs were older than those from GU reefs in all years in the Mackay region, in 1997 and 1998 in the Townsville region and in 1995-1997 in the Storm Cay region (Fig. 40), but younger or the same age otherwise.

Figure 40: Mean age of legal sized red throat emperor from unfished reefs zoned Marine National Park (MNP) and General Use (GU) in each region in 1995 - 1998. Error bars are Standard Errors.



Effects of management zone on sizes of red throat emperor were more variable than other effects. Legal sized red throat emperor were significantly longer on MNP reefs than on GU Reefs sampled during baseline surveys in the Mackay region, did not differ significantly with zone in the Townsville region and were longer on the GU reefs in the Storm Cay region (Zone x Region interaction $F_{2,12}=4.54, \alpha=0.034$; Fig. 41). Comparisons of mean lengths of fish between MNP Control and GU reefs were similarly variable among regions and also among years over the period 1995-98 (Zone x Region x Year interaction $F_{6,18}=2.32, \alpha=0.078$; Fig. 42). Consistent differences between zones were evident only in the Mackay Region, where red throat emperor were larger on average on MNP reefs than the single GU reef in these analyses (Fig. 42). In both the Townsville and Storm Cay regions, all possible

relationships among zones (MNP > GU, GU > MNP, MNP \approx GU) were evident over the four years (Fig. 42).

Figure 41: Mean Fork Length (FL) of legal sized red throat emperor from four unfished reefs zoned Marine National Park (MNP) and two General Use (GU) reefs averaged baseline surveys in 1995 and 1996 for each region. Error bars are Standard Errors.

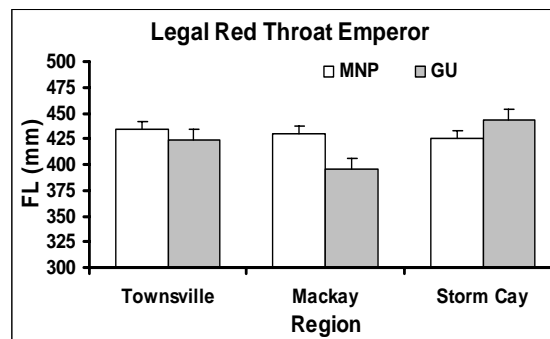
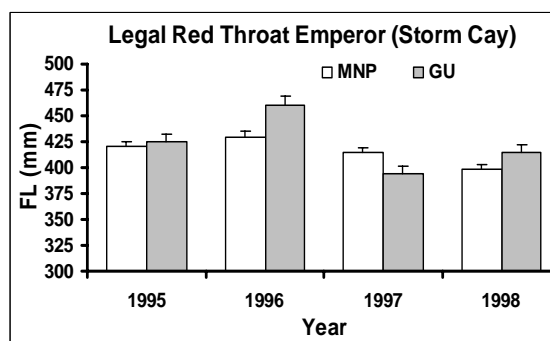
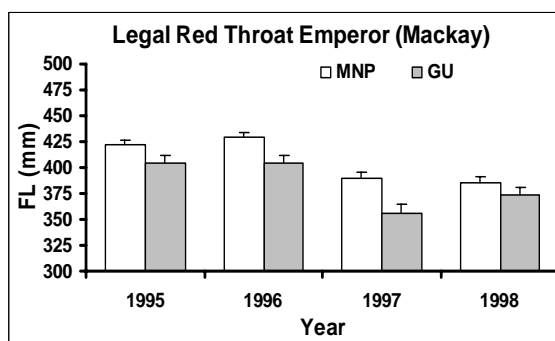
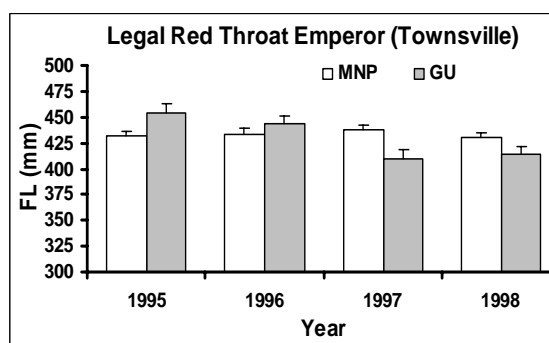


Figure 42: Mean Fork Length (FL) of legal sized red throat emperor from three unfished reefs zoned Marine National Park (MNP) and one General Use (GU) reef from each region in 1995-1998. Error bars are Standard Errors.



Mortality of red throat emperor was not compared between zones because there were low numbers of individuals and truncated age structures from GU reefs. These patterns alone qualitatively suggest considerably greater mortality on GU than on MNP reefs.

Effects on Non-catch Species

There were significant interactions between Zone, Region and Year for counts of small and large scarids from 1995-96 (small scarids - $F_{3,16}=5.27$, $\alpha=0.010$; large scarids - $F_{3,20}=4.64$, $\alpha=0.013$) and also for both groups over the period 1995-98 (small scarids - $F_{9,27}=2.26$, $\alpha=0.049$; large scarids - $F_{9,27}=2.27$, $\alpha=0.049$). The relationships between MNP and GU reefs, however, were not consistent over time in any region or over history, suggesting that counts were not predictably influenced by zoning history. For example, large scarids were significantly more abundant on MNP reefs in the Townsville Region in 1995 and Lizard Region in 1996 (Fig. 43) but at no other regions in either year (Fig. 43) nor at any regions in either 1997 or 1998 (Fig. 45). Conversely, these fish were more abundant on GU reefs than on MNP reefs in the Lizard Region in 1996 (Fig. 43).

Small scarids were significantly more abundant on GU reefs than on MNP reefs in the Lizard Region in 1995 and in the Townsville, Mackay and Storm Cay Regions in 1996 (Fig. 43), but were at similar densities in each region in the other year (Fig. 43). Erratic results also were apparent when the MNP Control reefs were compared with both GU reefs in each region over all years 1995-2000 (Zone x Region x Year interaction - $F_{15,40}=2.58$, $\alpha=0.009$; Fig. 44).

Figure 43: Mean densities of large scarids (Top) and small scarids (Bottom) from reefs zoned Marine National Park (MNP) and General Use (GU) in each region in 1995 (Left) and 1996 (Right). Error bars are Standard Errors.

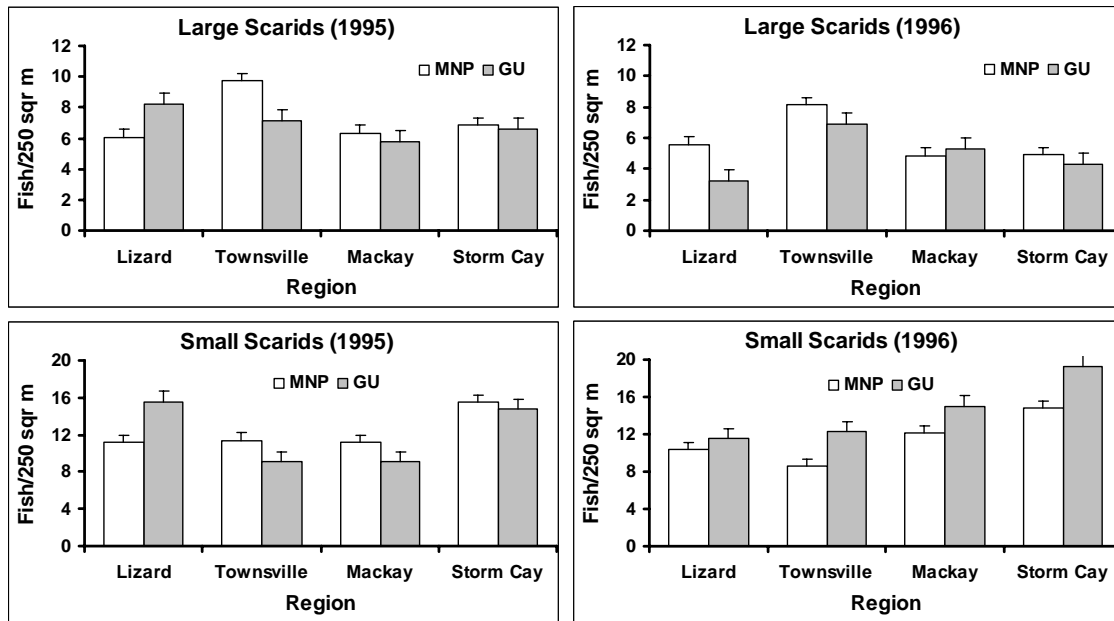
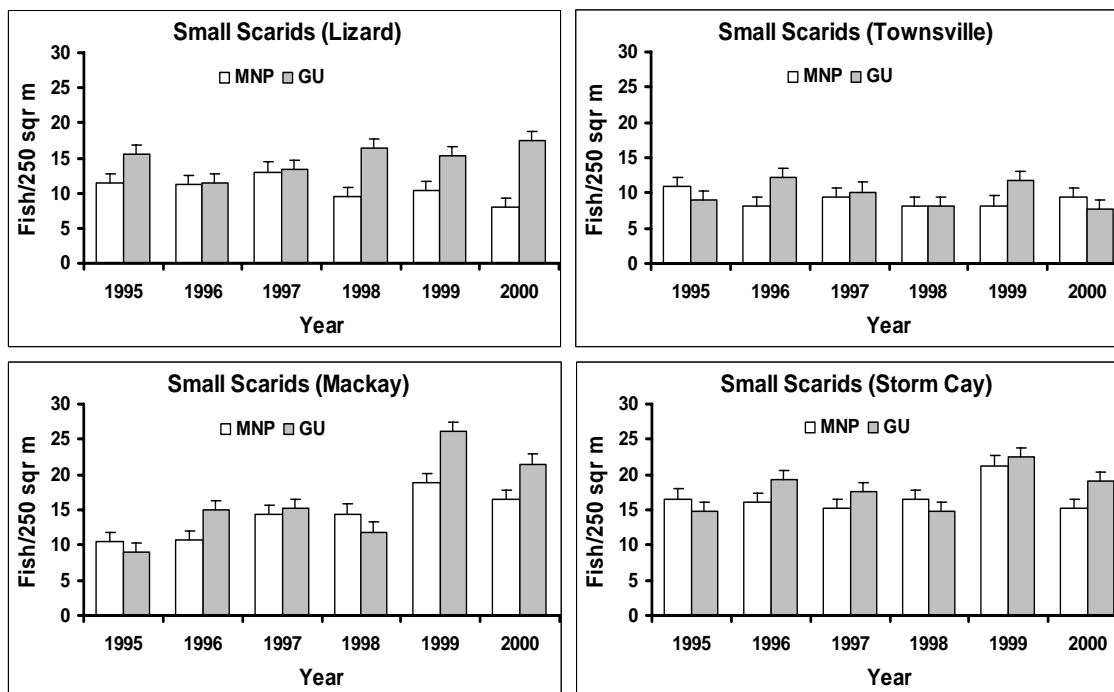


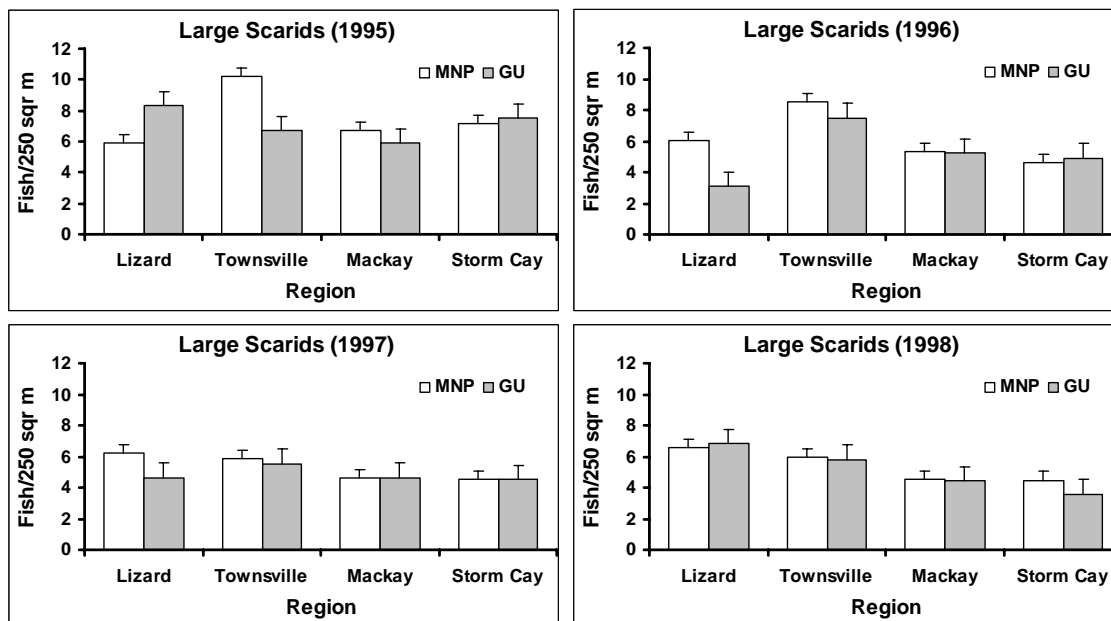
Figure 44: Mean densities of small scarids from Marine National Park (MNP) Control and General Use (GU) reefs in each region in 1995-2000. Error bars are SEs.



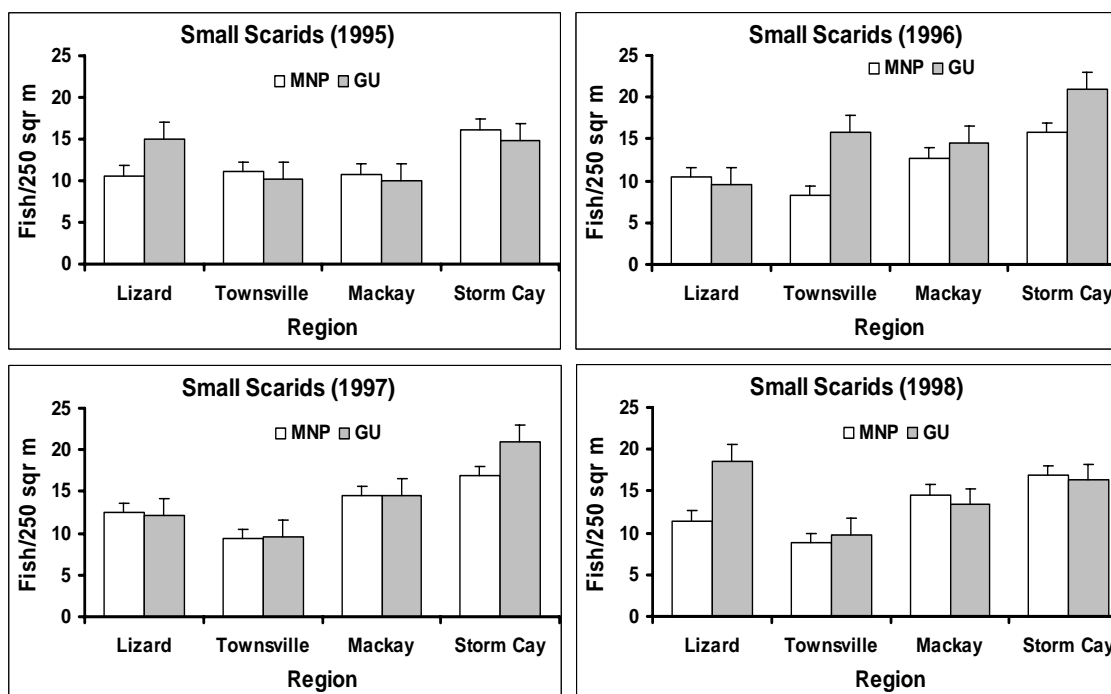
Similarly, in 1997, small scarids were more abundant on the General Use reef than on the three MNP reefs around Storm Cay but nowhere else, and in 1998 they were significantly more abundant on the GU reef in the Lizard Region but nowhere else (Fig. 45). It is noteworthy, however, that small scarids were never at significantly greater densities on MNP reefs than on GU reefs.

Figure 45: Mean densities of large scarids (A) and small scarids (B) averaged over Marine National Park (MNP) Control reefs and General Use (GU) reefs in each region in 1995, 1996, 1997 and 1998. Error bars are Standard Errors.

A) Large Scarids



B) Small Scarids



Effects of zoning on other non-harvested fish were evident for caesionids (Zone x Year interaction, 1995-98 - $F_{3,33}=2.26$, $\alpha=0.100$; Fig. 46) and grouped pomacentrids (Zone x Region x Year interactions, 1995-96 - $F_{3,16}=3.077$, $\alpha=0.058$; 1995-2000 - $F_{15,45}=1.71$, $\alpha=0.083$; Fig. 47). As with the scarids, there was no consistent Zone effect, although abundances of pomacentrids were generally significantly higher on closed than open reefs off Mackay and greater on open than closed reefs off Townsville. (Fig. 47).

Figure 46: Mean densities of caesionids on reefs zoned Marine National Park (MNP) and General Use (GU) averaged over regions in 1995, 1996, 1997 and 1998. Error bars are Standard Errors.

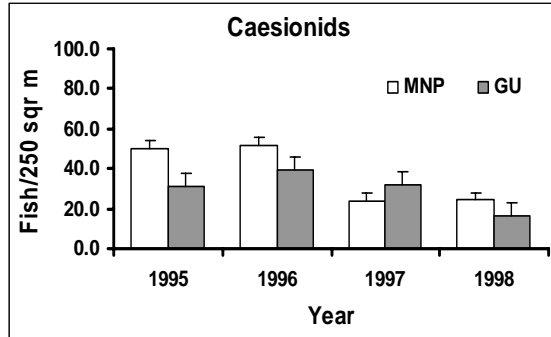
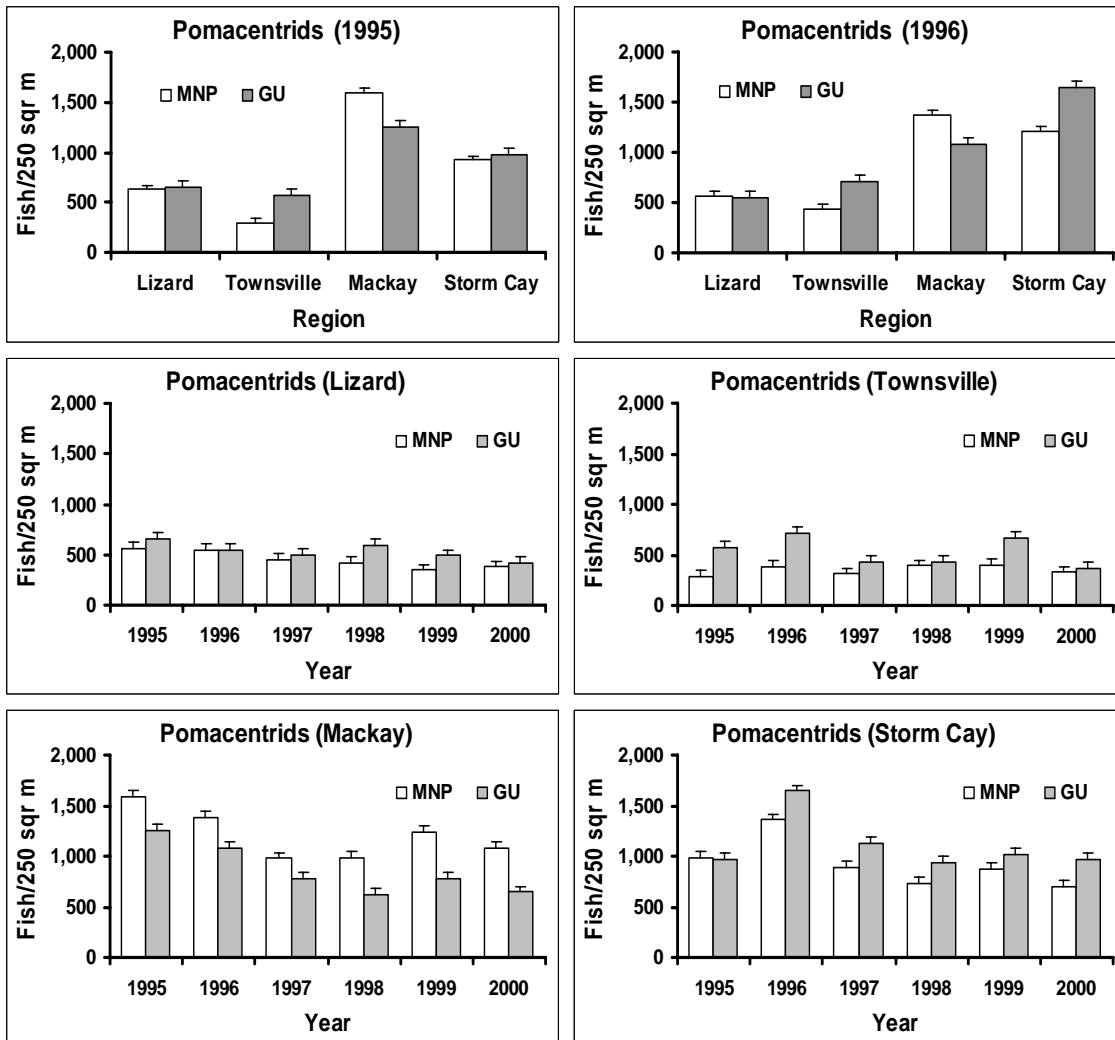


Figure 47: Mean densities of pomacentrids from Marine National Park (MNP) and General Use (GU) reefs in each region in 1995 (Top Left) and 1996 (Top Right) and in 1995-2000 for each region (Middle and Bottom). Error bars are Standard Errors.



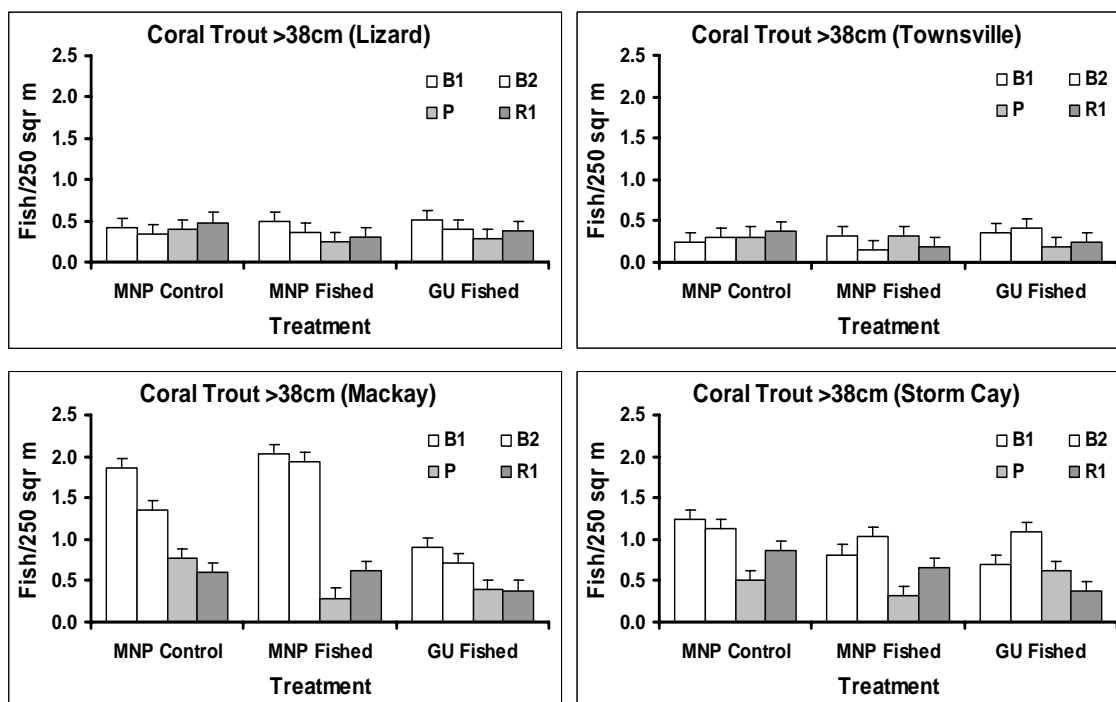
Effects of Fishing Manipulations

In general the manipulations of fishing and reef closures resulted in reduced counts, catch rates and sizes of legal size coral trout and increased catch rates of under sized coral trout in the southern most two or, in some cases, three regions. There was little evidence of impacts of the manipulations in the Lizard region. Although there were significant interactions between the years of manipulations and treatments for non-catch species, few of these interactions resulted from patterns in abundance that would be explained easily in terms of secondary impacts of fishing. Most of the apparent impacts of fishing were driven by impacts of fishing on the MNP reefs that were opened to fishing, with ambiguous evidence that the intended increases in fishing on GU reefs either occurred or was effective.

Effects on Legal Sized Coral Trout

When the two series of treatments were analysed together, effectively absorbing inter-annual variation in treatment effects into the residual variation, several patterns emerged related to the impacts of fishing. Counts of legal sized coral trout varied with Treatment and Treatment Year but this interaction varied with Region (Region x Treatment(Zone) x Treatment Year interaction – $F_{9,36}=2.238$, $\alpha=0.042$). Estimated population density in the Lizard region varied very little over the four treatment years (Baseline 1 and 2, Pulse, Rebuilding 1) and provided no statistically significant evidence of impacts of fishing (Fig. 48). Counts on all reefs in the Mackay and Storm Cay regions dropped significantly and dramatically during the year of Pulse fishing compared to those in baseline years and remained relatively low in the first year after reefs were closed for rebuilding (Fig. 48). These effects, however, were apparent in all treatments, including on control reefs, although the transients on the MNP Fished reefs were greater in magnitude than on the Control reefs (Fig. 48).

Figure 48: Underwater visual counts of legal sized coral trout (>38cm TL) on reefs in each of three treatments in the spring of four years in each region. Key: **B1** – Baseline year 1; **B2** – Baseline Year 2; **P** – Year of Pulsed fishing; **R1** – First year following closure of pulsed reefs to allow rebuilding of depleted stocks.



When broken down by zones, counts of legal coral trout varied among Treatment Years in consistent pattern over all regions on GU Fished reefs (main effect of Year - $F_{3,4}=11.7$, $\alpha=0.019$), abundance dropping by around 50% in the Pulse and first Rebuilding years compared with baseline years (Fig. 49). The regional variation in fishing effects above were driven by data from the MNP Reefs (Region x Treatment x Treatment Year interaction - $F_{9,24}=2.47$, $\alpha=0.037$), where there were no significant effects of fishing treatment or time on counts of legal sized coral trout in either the Lizard or Townsville regions, but strong effects off Mackay and around Storm Cay (Fig. 50). As noted above, although declines in counts concurrent with the manipulations in these regions occurred on both control and fished reefs, the effects were significantly stronger on the MNP Fished reefs, meaning that they changed from being either greater than (Mackay) or similar to (Storm Cay) MNP Control reefs in the year prior to Pulse fishing to being significantly less than the controls in the year of pulse fishing (Fig. 50).

Figure 49: Underwater visual counts of legal sized coral trout (>38cm TL) on General Use reefs in the spring of four years, averaged over regions. Key: **B1** – Baseline year 1; **B2** – Baseline Year 2; **P** – Year of Pulsed fishing; **R1** – First year following closure of pulsed reefs to allow rebuilding of depleted stocks.

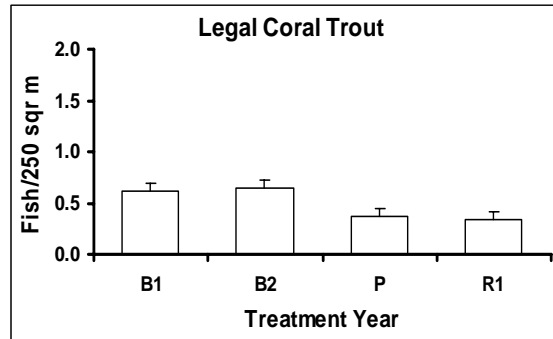
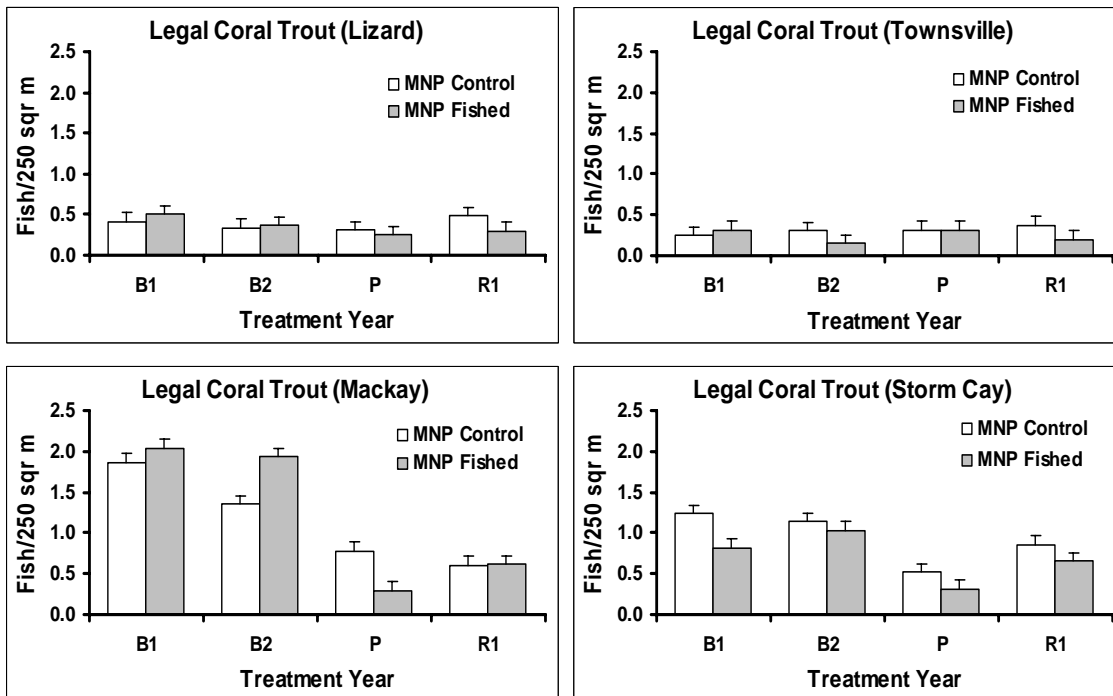
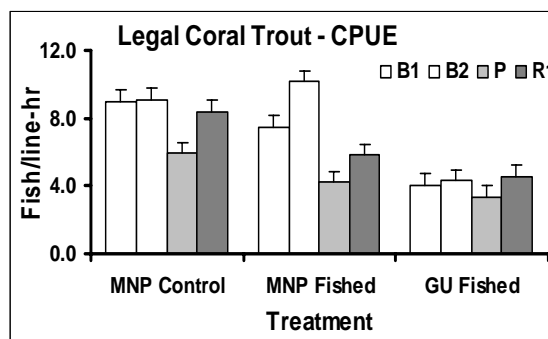


Figure 50: Underwater visual counts of legal sized coral trout (>38cm TL) on MNP reefs in each of two treatments in the spring of four Treatment Years in each region. Key: **B1** – Baseline year 1; **B2** – Baseline Year 2; **P** – Year of Pulsed fishing; **R1** – First year following closure of pulsed reefs to allow rebuilding of depleted stocks.



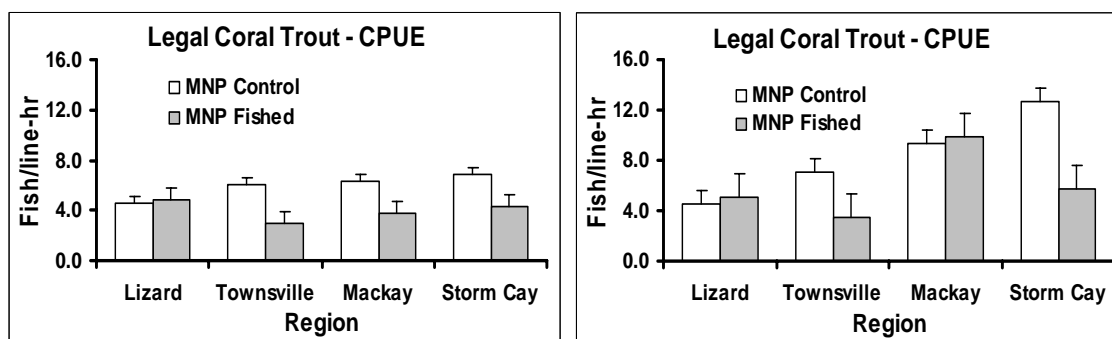
CPUE of legal sized coral trout varied significantly with Treatment in interaction with Treatment Year (Treatment(Zone) x Treatment Year interaction - $F_{3,45}=2.67$, $\alpha=0.059$) without interaction with Region (Region x Treatment(Zone) x Treatment Year interaction - $F_{9,45}=0.64$, $\alpha=0.759$). CPUE was significantly greater on both MNP Control and MNP Fished reefs than on GU Fished reefs during the baseline years, but dropped on both MNP treatments in the years of Pulse fishing (Fig. 51). The reduction in CPUE was significantly greater on the MNP Fished reefs than on the MNP Control reefs but was absent from the GU Fished reefs (Treatment Year - $F_{3,21}=1.59$, $\alpha=0.221$). CPUE on the MNP Fished reefs was reduced during the Pulse year to be similar to that on the GU Fished reefs but increased to be above the GU reefs in the first year of rebuilding (Fig. 51). CPUE on the MNP Control reefs 'rebounced' in the R1 year to be similar to baseline rates (Fig. 51).

Figure 51: CPUE of legal sized coral trout (>38cm TL) on reefs in each of three treatments averaged over regions in the spring of four years. Key: **B1** – Baseline year 1; **B2** – Baseline Year 2; **P** – Year of Pulsed fishing; **R1** – First year following closure to allow rebuilding of depleted stocks.



Despite the absence of a significant interaction between these effects and region, detailed examination of the effects indicated that they were precipitated mainly by strong effects of fishing on the MNP Fished reefs in Townsville, Mackay and Storm Cay regions in 1997-98 and in the Townsville and Storm Cay region in 1999-2000 (Fig. 52), with no significant effects in the Lizard region from either manipulation.

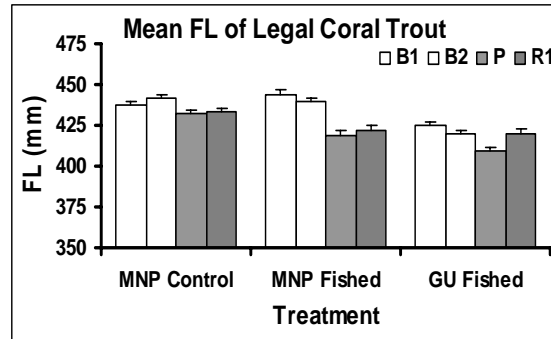
Figure 52: Catch per unit of effort (CPUE) of legal sized coral trout (>38cm TL) on Marine National Park reefs in each of two treatments in four regions averaged over the Pulse and first Rebuilding years for manipulations in 1997-98 (Left) and 1999-2000 (Right).



Mean lengths of harvestable coral trout also showed a Treatment(Zone) x Treatment Year interaction averaged across all regions ($F_{3,36}=6.04$, $\alpha=0.002$). Mean length of fish in the catch was reduced significantly by pulse fishing on MNP Fished reefs and decreased at the same time on MNP Control reefs, though by less than on the fished reefs (Fig. 53). Mean size also dropped during the Pulse year on the GU reefs, but was similar in the following

(Rebuilding) year to mean sizes during the baseline years (Fig. 53). The mean size on GU reefs was consistently less than that on MNP Reefs when the latter were not exposed to fishing (Fig. 53). Mean age of harvestable coral trout were not affected significantly by any treatment-related effects ($P>0.3$ in all tests).

Figure 53: Average fork length (FL) of legal sized coral trout (>38cm TL) on reefs in each of three treatments in the spring of four years in each region. Key: **B1** – Baseline year 1; **B2** – Baseline Year 2; **P** – Year of Pulsed fishing; **R1** – First year following closure of pulsed reefs to allow rebuilding of depleted stocks.



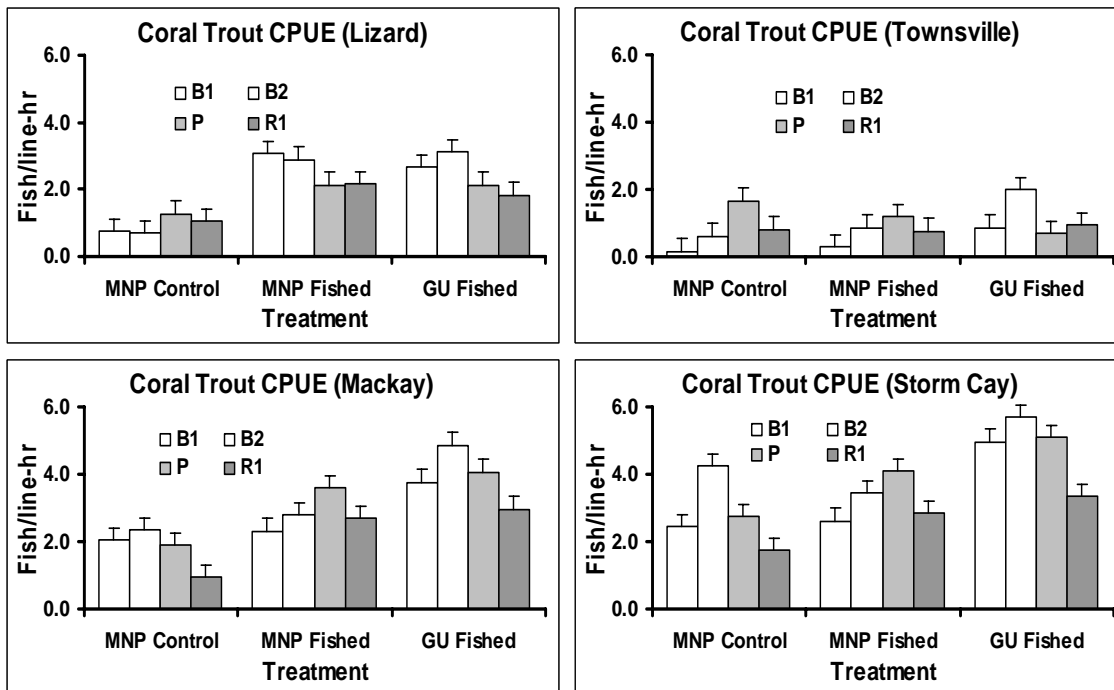
Effects on Under Sized Coral Trout

There were no significant interactions between Treatment and Treatment Year, either alone or in conjunction with other factors, for average length, age, or underwater counts of sub-legal sized coral trout ($P>0.1$ in all tests). CPUE of under sized coral trout, however, did vary with Treatment (Zone) and Treatment Year in interaction with Regions ($F_{9,48}=2.45$, $\alpha=0.022$). The effects were variable regionally and not consistently related to pulse fishing (Fig. 54-A). For example, catch rates of under sized coral trout were less on GU Fished reefs in all regions and on MNP Fished reefs in the Lizard region during and after the Pulse year than during one or both baseline years, but increased in the years of pulse fishing on MNP Fished reefs in the Townsville, Mackay and Storm Cay regions (Fig. 54-A). Similarly, CPUE of under sized coral trout on MNP Control reefs showed regionally variable changes in the manipulation years, ranging from being significantly less than in baseline years around Storm Cay, to no significant change from baseline years around Lizard Island and off Mackay, to a significant increase over baseline years off Townsville (Fig. 54-A).

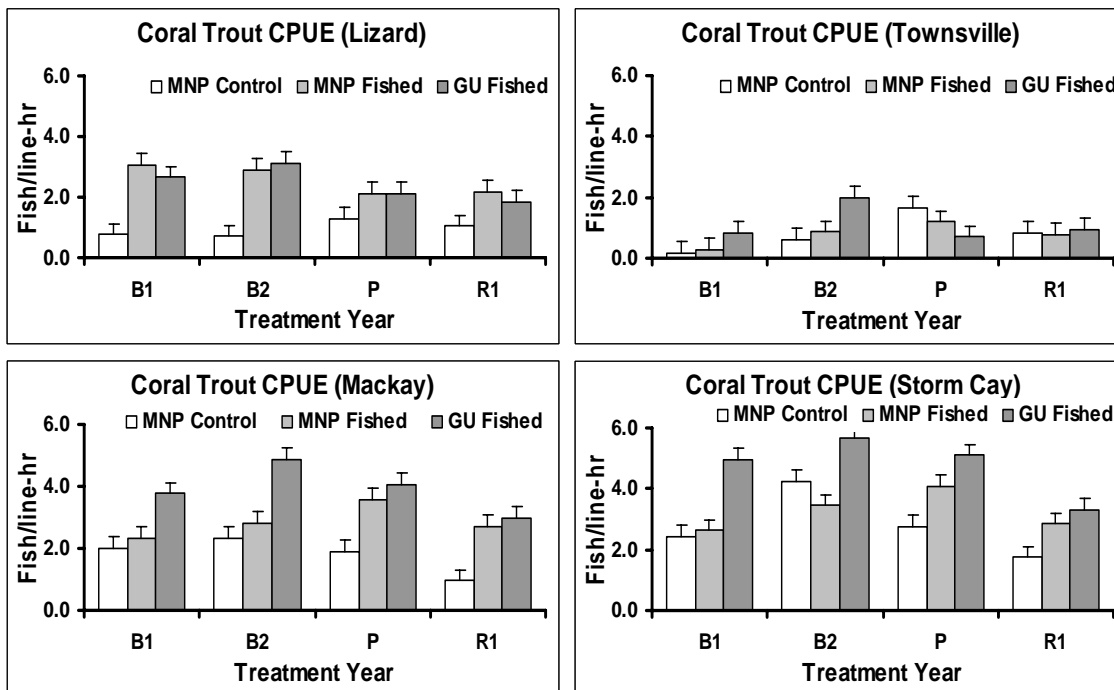
Some interesting relatively consistent (over years) contrasts in CPUE existed among reefs in different treatments (Fig. 54-B). Catch rates of under sized coral trout on General Use reefs were significantly greater than on MNP Control Reefs in all years in the Lizard, Mackay and Storm Cay regions and in one year off Townsville. In the Lizard region, CPUE also was significantly greater in all years on MNP Fished Reefs than on MNP Control reefs, and did not differ in any years between MNP Fished and GU reefs (Fig. 54-B). In the Mackay and Storm Cay regions, where impacts of pulse fishing on MNP Fished reefs were greatest on legal sized coral trout (above), catch rates of under sized coral trout on the MNP Fished reefs jumped from being similar to those on MNP Control reefs during the baseline years to being more similar to those on the GU reefs in the Pulse and subsequent years (Fig. 54-B).

Figure 54: Catch per unit of effort of under sized coral trout (<38cm TL) on reefs in each of three treatments in the spring of four treatment years in each region. *Key: B1 – Baseline year 1; B2 – Baseline Year 2; P – Year of Pulsed fishing; R1 – First year following closure of pulsed reefs to allow rebuilding of depleted stocks.* In **A)** the data are arranged to illustrate effects of pulse fishing, whilst in **B)** data are arranged to illustrate static relationships among reefs.

A)



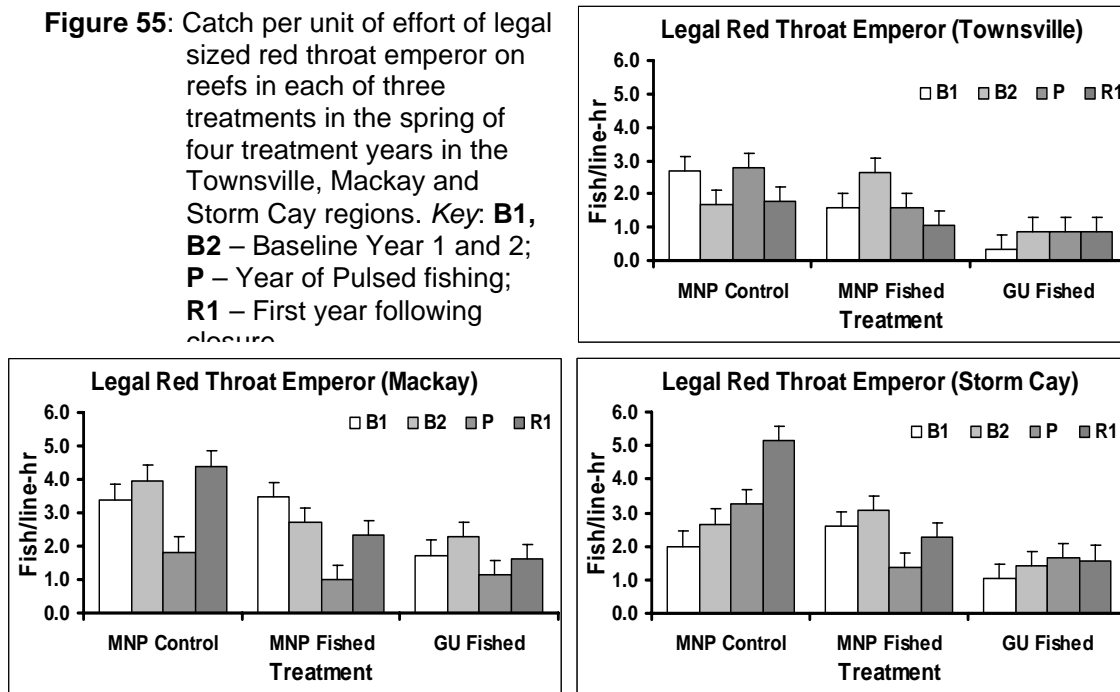
B)



Effects on Red Throat Emperor

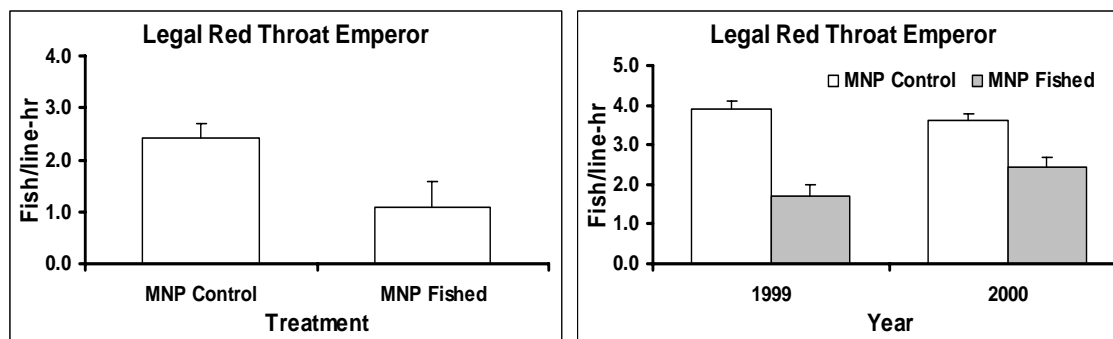
There was a significant interaction between fishing Treatment(Zone), Treatment Years and Region for red throat emperor ($F_{6,33}=2.99$, $\alpha=0.019$). The apparent impacts of the fishing treatments were restricted to MNP reefs, however, there being no significant effects for General Use Reefs ($F_{3,15}=1.07$, $\alpha=0.392$). When averaged over the two years of manipulations, CPUE of legal sized red throat emperor on MNP Fished reefs was generally lower than on MNP Control reefs during and after the pulse fishing years, despite having been similar to or greater than those on control reefs in baseline years (Fig. 55). Further, increases in CPUE in Pulse and Rebuilding years on MNP Control reefs were damped or absent on the MNP Fished reefs (Fig. 55).

Figure 55: Catch per unit of effort of legal sized red throat emperor on reefs in each of three treatments in the spring of four treatment years in the Townsville, Mackay and Storm Cay regions. Key: **B1**, **B2** – Baseline Year 1 and 2; **P** – Year of Pulsed fishing; **R1** – First year following closure



When contrasts between MNP Control and MNP Fished reefs were compared separately for each treatment event (1997-98, 1999-2000), significant differences consistent with expected impacts of fishing were evident in both pulses in all regions (Main effects of Treatment, 1997-98 - $F_{1,8}=5.59$, $\alpha=0.046$; Treatment x Year interaction, 1999-2000 - $F_{1,3}=8.34$, $\alpha=0.063$; Fig. 56). In the 1997-98 period, effects were consistent over years (Pulse and Rebuild 1) and all regions, whilst in the second manipulation period, CPUE on MNP Fished reefs had increased slightly in the Rebuild year compared to the year of Pulse fishing, although in both years CPUE from MNP Fished reefs was significantly lower than on MNP Control reefs in all regions (Fig. 56).

Figure 56: Catch per unit of effort (CPUE) of legal sized red throat emperor on Marine National Park reefs in each of two treatments in four regions averaged over the Pulse and first Rebuilding years for manipulations in 1997-98 (left) and separately for the two years of the second manipulations (1999-2000) (right). Error Bars are Standard Errors.



Whilst there were significant inter-annual, regional and zoning effects on mean age and size of red throat emperor, there were no significant interactions between Treatment and Treatment Year or patterns in age or size among treatment reefs in either manipulation period that would signal demonstrable impacts of the pulse fishing treatments ($P > 0.1$ in all tests).

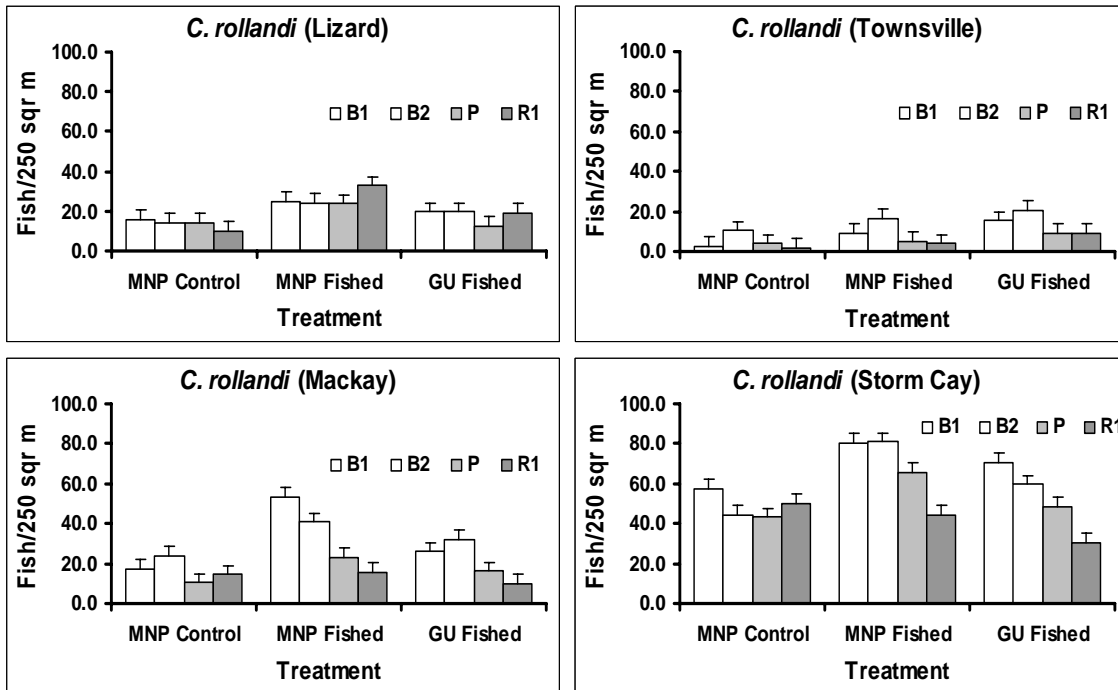
Effects on Non-catch Species

Interactions between Treatment(Zone) and Treatment Year were significant, either alone or in interaction with Region, for several non-harvest species that might reflect secondary effects of removal of higher predators by fishing. There was little consistency among such potential effects, however, and the interactions often appeared to be driven more by inter-annual variations than treatment effects. For example, there was a significant Region x Treatment(Zone) x Treatment Year interaction for *Chrysiptera rolandi* ($F_{9,48}=3.19$, $\alpha=0.004$), with no significant variations over time in abundances on any groups of reefs in the Lizard region and abundances in other regions generally lower during the Pulse and re-building years on treatment reefs (MNP Fished and GU Fished) than during the immediately preceding baseline year (B2) (Fig. 57). Similar reductions were apparent on the MNP Control reefs, however, in Townsville and Mackay regions, though not in the Storm Cay region (Fig. 57).

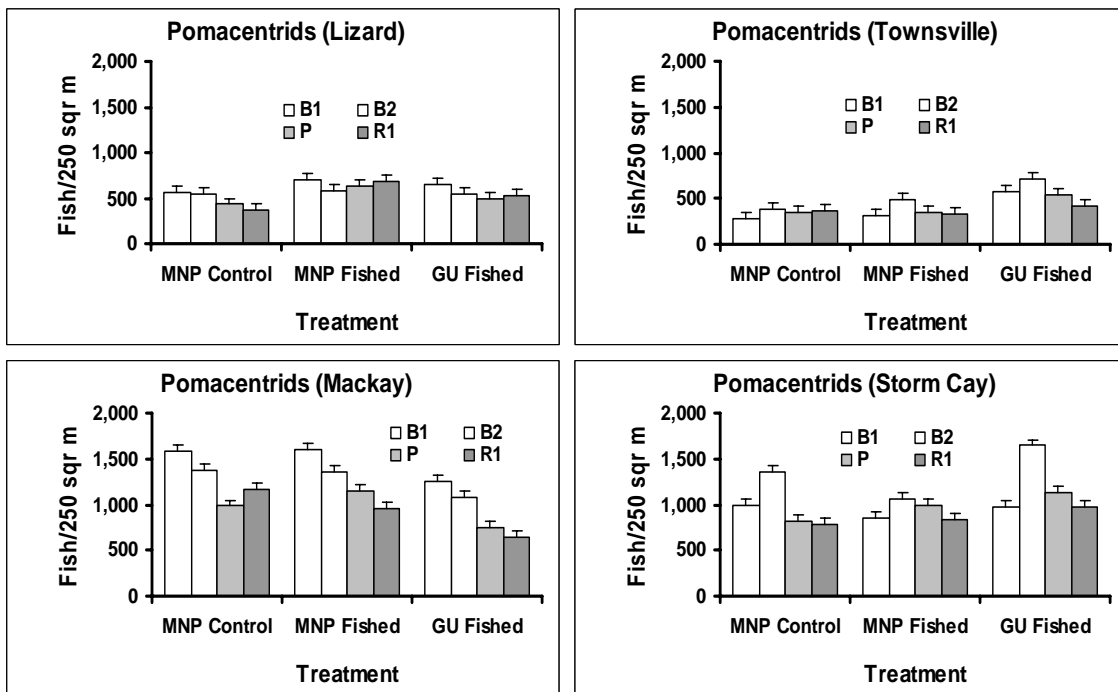
Similarly, abundances of an aggregate of pomacentrid species varied little over time in any treatment in the Lizard and Townsville regions or on MNP Fished reefs near Storm Cay, but showed consistent declines in abundance on MNP Fished and GU Fished reefs off Mackay from the first to the last years of analysed (Fig. 57; Region x Treatment(Zone) x Treatment Year interaction - $F_{9,45}=2.20$, $\alpha=0.040$). Thus, these patterns were not consistently or conspicuously related to the years in which fishing treatments were imposed.

Figure 57: Underwater visual counts of *Chrysiptera rollandi* (A) and other pomacentrids (apart from *P. moluccensis* and *A. curacao*) (B) on reefs in each of three treatments in the spring of four years in each region. Key: **B1** – Baseline year 1; **B2** – Baseline Year 2; **P** – Year of Pulsed fishing; **R1** – First year following closure of pulsed reefs to allow rebuilding of depleted stocks.

A) *Chrysiptera rollandi*



B) Other Pomacentrids



Small scarids also varied with Treatment Year in interaction with Treatment(Zone) and Region ($F_{9,39}=2.01$, $\alpha=0.064$), but again the patterns of variation were not obviously related to

those expected if they had been generated by the fishing treatments (Fig. 58). *Pomacentrus moluccensis* varied with Treatment(Zone) and Treatment Year consistently over regions ($F_{3,54}=2.61$, $\alpha=0.061$), but that interaction arose because a) densities on MNP Control reefs in the Pulse year were less than on MNP Fished reefs in the same year whilst in the following year, densities were lower on average on General Use reefs than on MNP Control or MNP Fished reefs (Fig. 59). Densities during the baseline years were similar across all reefs, irrespective of treatment, and dropped in all treatments by similar amounts (~20%) in the Pulse and following years (Fig. 59).

Figure 58: Underwater visual counts of small scarids (<30cm TL) on reefs in each of three treatments in the spring of four years in each region. Key: **B1** – Baseline year 1; **B2** – Baseline Year 2; **P** – Year of Pulsed fishing; **R1** – First year following closure of pulsed reefs to allow rebuilding of depleted stocks.

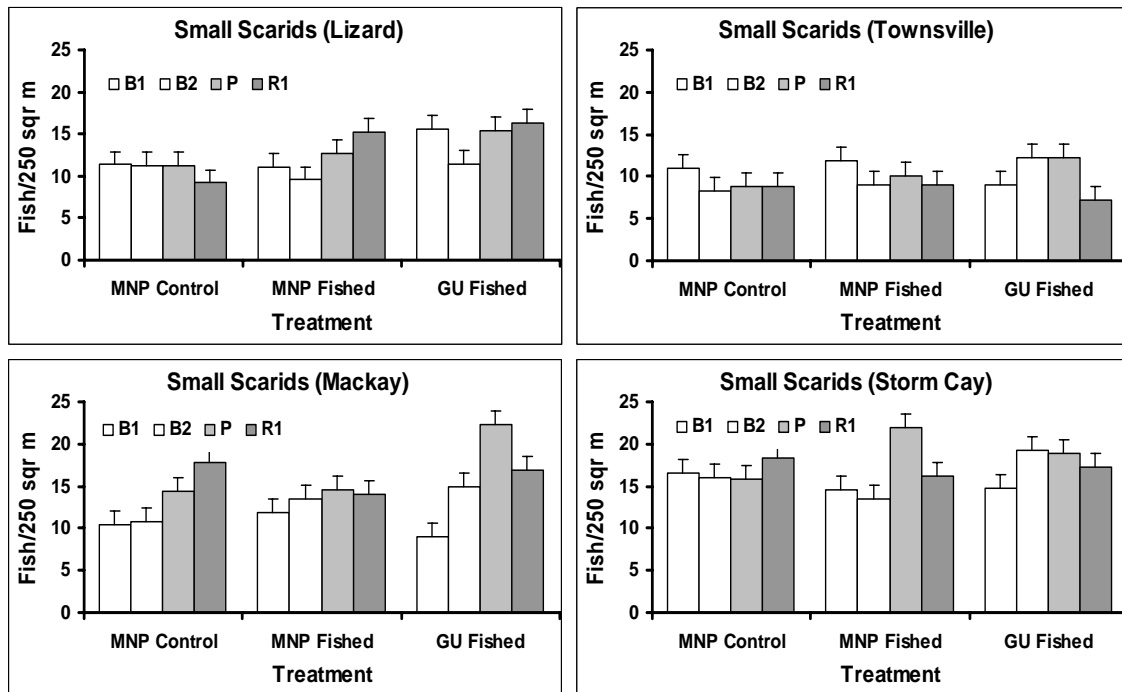
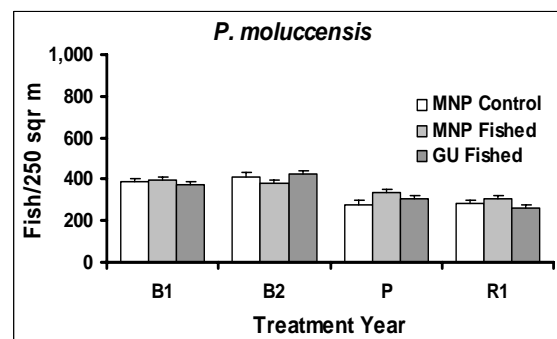


Figure 59: Underwater visual counts of *Pomacentrus moluccensis* on reefs in each of three treatments averaged over all regions in the spring of four years. Key: **B1**, **B2**– Baseline years 1 and 2; **P** – Year of Pulsed fishing; **R1** – First year following closures.



Discussion

We have demonstrated through the Effects of Line Fishing (ELF) Experiment the capacity for line fishing on the Great Barrier Reef (GBR) to impact significantly the status and characteristics of target reef fish stocks. Such impacts, though expected (Johannes and Riepen 1995, Polunin and Roberts 1996, Russ *et al.* 1995, 1996, Sadovy and Vincent 2002), can be manifest over a relatively short time (one year in this case) to the extent that the characteristics of populations previously protected to some extent from fishing can be rendered indistinguishable from populations that have been harvested for decades (Alcala and Russ 1990, Russ and Alcala 1989, Russ 2002). We cannot demonstrate, however, how closely the depletion through pulse fishing reflects the impacts of fishing an unharvested (virgin) resource since we cannot be certain how effective the notional closures to fishing have been and how close to virgin were the populations on the MNP reefs. No data exist from which to assess whether recruitment and survivorship of coral trout and red throat emperor since the MNP reefs were closed to fishing (late 1980s) have been sufficient to rebuild populations to their virgin status.

Implementing the Experiment

The magnitudes of responses of the Reef Line Fishery (RLF) to small amounts of new fishing ground (the opening of MNP-Fished reefs) varied regionally, largely in proportion to the existing regional distribution of effort. Fishers in general seemed unwilling to travel significantly beyond their historical fishing range, even with the prospect of harvesting resources that were believed to be more abundant (by virtue of being closed to fishing), in spite of widespread belief that some poaching of those resources had already occurred. Thus, our capacity to precipitate changes in fishing pressure and consequent changes in the resource was limited by the availability of effective effort in each of the regions. This meant that there was negligible, if any, impact of the manipulations in the Lizard Region in either round of reef openings, largely due to the low numbers of vessels operating in that region.

Further, we realised only limited success in increasing fishing effort on reefs already open to fishing. Whilst this might not be surprising, it signifies a practical problem with the implementation of experiments such as the ELF Experiment. Invariably, the success of such large-scale experiments with a resource management focus will require the support and participation of industry sectors. That participation typically will depend on two things: clear benefits to users of participating in the experiment (in this case, to increase fishing on selected reefs), and the credibility of the proposed manipulative procedures, especially the degree to which induced changes in harvest regimes can be considered believable and relevant to existing or prospective industry practices. It is clear from the poor response to entreaties to 'fish harder' on existing open reefs that the prospect of improved information for management of the fishery constituted insufficient incentive for participation. Additional, probably economic, incentives would have been required to produce more substantial treatment effects on the General Use reefs. The purchase of dedicated fishing effort on target reefs was not supported initially by users because it was considered to represent uncharacteristic fishing behaviours that might precipitate results of dubious relevance to management of the real fishery. Moreover, the financial subsidy of fishing sufficient to concentrate effort on reefs with diminishing resources and impose desired treatment effects would have been prohibitive, especially in the context of the rapid and sustained increases in beach prices for live reef fish over recent years. Such obstacles will likely exist for any fishery-scale manipulative experimental approaches to management research and will be a major challenge in the implementation of any future research of the type we have done.

Biomass Estimates and Future Monitoring Procedures

Despite the lack of success in pulse fishing open reefs, the ELF experiment did manage to achieve significant effects of fishing on the opened MNP reefs. This successful treatment has provided both additional inferential strength to our assessments of the utility of area closures on the GBR and the only current estimates of reef-specific biomass and the utility of underwater visual surveys and line fishing catch surveys for future monitoring of coral trout

populations. Whilst further refinement of the depletion estimators is clearly warranted (see Appendix A), work so far indicates that the completion of the ELF Experiment will yield our first reasonable estimates of biomass on at least a subset of the reefs of the GBR. Extrapolating such estimates to the entire GBR, however, will hinge on improving our knowledge of the extent of reef habitat (see following section of Management Strategy Evaluations) and better establishing the relationship between existing maps of reefs and the biomass on them. These are important directions for future research.

Results from the experimental data so far indicate that CPUE data from catch surveys are likely to be considerably more responsive to changes in abundance than underwater visual surveys. Catch surveys have the potential to sample over the entirety of the fished habitat, whereas UVS for the most part is restricted to habitat less than 12m deep. Further, catch surveys provide additional and more precise data on the sizes and ages of harvested fish that are not available from UVS. Alternatively, the UVS can provide a broader range of ecological data that will not be provided from catch surveys. Further work is required before definitive conclusions can be drawn about the advantages and disadvantages of the two monitoring methods for tracking populations of harvested species, but it is clear that non-destructive sampling (counting) cannot provide the full suite of data necessary for resource assessments. Where area closures are used for resource management, this means that some destructive sampling of populations within closed areas will be required to adequately measure the performance of the area closure strategies.

Effects of Marine Park Zoning

Despite uncertainty about the degree of compliance with area closures to fishing, we have demonstrated that *P. leopardus* and *L. miniatus* on at least some of the reefs closed to fishing for over a decade were significantly more abundant, larger and older on average than on nearby reefs that have always been open to fishing. Such contrasts have been reported widely on coral reefs elsewhere, especially where fishing intensity is high (Alcala and Russ 1990, Bohnsack 1994, Polunin and Roberts 1993, Russ and Alcala 1989, see Russ 2002 for a thorough review), but similar results have not been so frequent on the Great Barrier Reef (Mapstone *et al.* 1999, Russ *et al.* 1995, 1996). Patterns such as these often are taken *prima facie* as evidence of the effectiveness of Marine Protected Area strategies in protecting marine resources from harvest (Russ 2002). In a recent review, however, Russ (2002) pointed out that this inference is logically flawed because areas that can be or are considered desirable to be closed to exploitation may be different *a priori* to those areas that are not closed to exploitation. Two main circumstances might result in such bias in the choice of areas for closure to harvest. First, where there is great social or economic dependence on fishing, it is likely that only areas that are little used (and so lightly exploited) will be available socially or politically for closure. Second, in choosing areas for protection from harvest, and perhaps other impacts of use, it is likely that sites that are considered to be 'special' in one or more respects will be favoured over 'average' sites. Thus, areas with greater diversity, more or larger fish, where animals aggregate at times, etc. may be more likely to be included in MPA networks than other sites. Moreover, it is more often the case than not that data about the status of closed areas relative to surrounding areas is not available from prior to closure or even for several years after closure. This is generally the case for various zones in the GBR Marine Park (Mapstone *et al.* 1999), including the reefs in the ELF Experiment. As Russ (2002) points out, for all these reasons simple post-hoc, retrospective comparisons of areas open and closed to fishing may provide misleading information about the effectiveness of MPAs as either conservation or harvest-protection management strategies.

Our manipulations of the closure status of reefs, however, has provided a considerably sounder basis from which to infer the effectiveness of the 'no-take' zoning strategies used for conservation management on the GBR, even though we do not have data prior to declaration of these zoning strategies for most of the reefs in the ELF Experiment. Our observations of rapid changes in population characteristics of the primary target species of the Reef Line Fishery when closed areas were opened to fishing adds weight to the inference that those reefs would have been similar to surrounding open reefs had they not been closed to fishing.

Thus, irrespective of at least some level of infringement, zoning reefs as 'no take' apparently has protected harvest species to a considerable extent from the impacts of line fishing.

In addition to the conspicuous impacts of open-access fishing on protected populations, there are early indications of increasing abundance and sizes of *P. leopardus* and *L. miniatus* on reefs protected from fishing, either after pulse fishing or following decades of exposure to fishing. Verification of such growth of populations and individuals through the remaining period of the ELF Experiment would significantly strengthen the inference that area closures on the southern GBR have been effective in protecting stocks from impacts of fishing.

Regional Variation

Regional replication of experimental treatments at whole-reef scales is an essential component of the ELF Experiment. This regional structure in the design has proved particularly important because it has enabled us to illustrate significant regional variation in both the responses of fishing fleets and fish stocks to past zoning strategies and opening and closing reefs to fishing. The above arguments about the effectiveness of zoning strategies apply convincingly only to the southern two (Mackay and Storm Cay) regions in which the ELF Experiment was done. Arguably, similar, though weaker, inferences can be made also in the Townsville region. There was considerably less, if any, substantive evidence for either the impacts of fishing or effects of historic reef closure in the Lizard Region, however. Three hypotheses can be posited to explain this regional variation and we have only circumstantial evidence and argument to discriminate among them.

First, it might be argued that rates or extent of infringement of closed areas have been considerably greater in the northern GBR than in the southern GBR, thereby eliminating the 'refuge' effects of (notionally) closing reefs to fishing. This seems unlikely for a number of reasons. The distribution of fishing effort and catch from all sectors of the RLF is substantially biased toward regions south of Cairns (Mapstone *et al.* 1996, Green *et al.* in prep, Higgs 2002), with fishing effort regularly being approximately 2-8 times greater in the southern regions than near the experimental reefs around Lizard Island. Thus, the rate of infringement (proportion of fishing days spent 'poaching') would have to be several times higher in the Lizard Region than in the southern regions to generate even similar impacts of infringement in closed areas. This seems unlikely in part because of the proximity of the ELF experimental reefs in the Lizard region to Lizard Island, where there is a permanent presence at the Lizard Island Research Station and Lizard Island Resort. Combined with the daily flights to and from Lizard Island that traverse the experimental reefs, this would be a significant deterrent to infringement on the closed experimental reefs. Further, aerial surveillance to the north of Cairns is the most frequent along the GBR, for reasons primarily unrelated to fishing. The reef matrix is at its narrowest in the northern GBR (Fig. 1), meaning that surveillance intensity over any given reef is likely to be considerably greater than on reefs south of 18°S where the reef matrix is significantly wider, more extensive and farther off-shore and aerial surveillance relatively less frequent (GBRMPA, unpub. data). Taken together, these circumstances would be expected to make infringements of area closures in the Lizard region less likely than on the experimental reefs in other regions.

Second, it might be argued that the population dynamics of coral trout in the Lizard Region (red throat emperor do not usually occur there) make the detection of differences between open and closed reefs more difficult than in the southern regions. Certainly, historical and current evidence is that population densities and annual recruitment of coral trout in the northern GBR are considerably less than in southern regions. These features alone would render the statistical power to detect differences among management zones less than where fish were more abundant, given the same sampling intensity (Mapstone *et al.* 1998c). Lower rates of recruitment would diminish the potential for protected reefs to accumulate large populations of otherwise harvested fish and would also diminish the capacity for harvested populations to re-build, so perpetuating the effects of annual harvest. Conversely, where recruitment was on average greater, the rate and amount of accumulation of fish on protected reefs would be accelerated as would be the rate of recovery from depletion on fished reefs. It is not categorically clear what would be the balance between these apposite consequences of

low (or high) recruitment or whether the net effect would diminish or enhance contrast between fished and protected reefs. It is likely, however, that the distribution of commercial fishing effort is in part a response to the capacity of reefs to sustain harvest year after year, substantially driven by recruitment. For example, it is a widely touted anecdote in the commercial fishery that the northern reefs are 'far less productive' and 'take longer to bounce back after being fished' than reefs in the southern GBR. If this is the case, then the effects of high or low recruitment might favour the exaggeration or diminution respectively of the effects of protection from harvest.

Third, it might be hypothesised that the absence of conspicuous differences between closed and open reefs and only weak or negligible effects of fishing manipulations in the Lizard Region simply might be the product of relatively low levels of fishing and correspondingly slight impacts of fishing where it occurs. Certainly, the correlation between the magnitude of impacts of fishing and differences between historically open and closed reefs we have documented would be consistent with this hypothesis.

In reality, it seems likely that all three hypotheses contribute to the regional differences in the apparent 'effectiveness' of reef closures. Importantly, however, this variation and the multiplicity of potential explanations for it highlight some key issues in assessing the effectiveness of area closure strategies and the deficiencies of seeking to demonstrate 'effectiveness' only in retrospect. Four things will be crucial to avoiding such ambiguities in the future assessments of zoning strategies on the GBR, such as those being introduced in the Representative Areas Program. First, it is critical that the objectives for which Marine Protected Areas (MPAs) are being established are clearly articulated. Second, it is important to identify those (usually human) impacts from which the closed areas are being protected and which will be most likely to precipitate contrast between MPAs and surrounding areas through their continued impacts on unprotected areas. Third, it is necessary to assess the status of the closed and surrounding areas prior to the introduction of exclusions, to document their relative status and to evaluate the likely existing and potential future impacts of designated activities. Finally, it will be necessary to establish regular monitoring of key variables that will inform assessments of both changes or stasis in status of the protected and unprotected areas and the dynamics of the impacting activities, the removal of which is expected to precipitate changes in state in the protected areas. Without each of these components, correctly interpreting future assessments of the utility of MPA strategies will be compromised and potentially result in erroneous conclusions that such strategies are ineffective (a Type II error). Had we considered only the reefs in the Lizard Region, for example, we might have come to just such a conclusion (Mapstone *et al.* 1998b).

Indirect Effects of Fishing

Contrasts between reefs historically open and closed to fishing were evident on the catch rates of both legal and under sized coral trout. Differences in catch rates and fishery-independent estimates of density of legal sized fish would be expected if fishing was sufficient to generate impacts on abundance and area closures were substantially respected. Differences in abundance indices of sub-legal fish, however, might indicate indirect effects of fishing due to post-release mortality. The differences that we observed, however, involved greater abundances of sub-legal fish in areas open to fishing, contrary to expectation if the indirect impacts of fishing were via post release mortality. Ayling *et al.* (1991) and Mapstone *et al.* (1997) suggested that there was fairly consistent evidence of small increases in abundance of sub-legal *P. leopardus* on fished compared with unfished reefs over many years of underwater visual surveys. They hypothesised that these patterns might reflect net reduction in mortality of juvenile coral trout because of reduced cannibalism on fished reefs. St. John (1996) demonstrated that *P. leopardus* are indeed cannibalistic, but the rates of cannibalism suggested by her extensive sampling of coral trout diets seemed relatively slight and probably inconsequential.

Whilst we found some evidence of relatively strong contrast between historically fished and protected reefs that would be consistent with a predation-release hypothesis, our data also suggest that perceived abundances of juvenile and under sized coral trout might be influenced

by another process. We found significant increases in catch rates of under sized coral trout on those reefs where pulse fishing substantially reduced catch rates and counts of legal sized coral trout. These relatively large increases occurred within a year of reefs being opened to fishing, making it unlikely that they could have arisen from increased survivorship alone. Direct observation of coral trout behaviour around baited hooks suggests that there is a size-based hierarchy in feeding behaviour, especially when many fish are present at a bait. Large *P. leopardus* have been observed actively chasing smaller coral trout away from baits, even if the larger fish does not take the bait immediately. Further, preliminary analyses of catch sequence data indicate that the larger fish taken on a hang are taken earlier in the hang than smaller coral trout. Thus, the dynamics in catch rates of smaller coral trout may be in part a product of interactions with larger coral trout, mediated by the density of the larger fish. Similarly, it may be that juvenile coral trout behave more cryptically and so have less visibility to divers when large fish are abundant, thus contributing to the perception of a compensatory response in juvenile survival to harvest of larger fish.

Whilst it has been shown that coral trout do not move significantly among reefs after settlement, it is generally believed that red throat emperor probably do migrate among reefs post-settlement (Brown *et al.* 1994, Williams *et al.* in press, W. Sumpton and I. Brown unpub. data, A. Williams unpublished data). Movement among reefs subject to different zoning would be expected to diminish the effectiveness of area closures as harvest refuges, with the diluting of the effect increasing with greater migration among reefs (Walters and Sainsbury 1990, Russ 2002, Mapstone *et al.* 1998b, 1999). We found that catch rates and mean age of red throat emperor on MNP reefs often were significantly greater than on GU reefs and that opening MNP reefs to fishing did significantly reduce abundances to values similar to those on reefs that had always been open to fishing. The existence of differences between reefs in different zones suggests that whatever inter-reef migration of red throat emperor might occur, it is insufficient to counter the effects of closing reefs to fishing, or to compensate for the effects of harvest on open reefs. Williams *et al.* (unpublished data) suggest that red throat emperor on the GBR might undergo extended inter-regional migration over several years, effectively diffusing outwards from centres of high recruitment. Such large-scale movement would involve reef to reef migration, but our data suggest that the residence times on each reef are likely to be relatively long (years rather than months), thus facilitating the accumulation of biomass on reefs where fishing mortality is low or absent.

We saw little evidence of immediate responses to the ELF Experiment in abundances of prey species of the common coral trout. This is an expected short-term result, since the effects of the experimental manipulations would be expected to be mediated through changes in predation on such species and, thus, unlikely to be manifest immediately. Taxa such as pomacentrids and scarids that are known to be relatively long-lived (Choat and Axe 1996, Choat *et al.* 1996, Choat and Robertson 2002, Doherty and Fowler 1994, Fowler and Doherty 1992, Mapstone 1988) might take some years to exhibit any responses to diminished or increased predation, if at all.

We also found little evidence for consistent differences in abundances of prey taxa between management zones, even where fishing pressure has been highest and impacts of fishing on target species apparent (Townsville, Mackay and Storm Cay regions). Though differences occurred, in some case consistently, they generally varied with region and were both consistent with a predation-release response and contrary to such an indirect effect. The ambiguity of these patterns may have arisen in part from strong and persistent regional patterns in abundances of prey species (Mapstone *et al.* 1999, Oliver *et al.* 1985, Sale *et al.* 1984) and because of considerable inter-annual variation in abundances. Both of these effects were likely to be substantially driven by recruitment processes (Doherty 1987, 1991, Doherty *et al.* 1984) and are likely to both complicate and make more difficult to detect secondary effects of the Reef Line Fishery.

Management Strategy Evaluations

Introduction

Management of the harvest of natural resources typically is reactive rather than proactive (Hilborn and Walters 1992) and conservative rather than adaptive (Walters 1986). Management planning and regulatory actions for fisheries usually follow the initial exploitation of targeted resources and adjustments to regulatory measures are made most often in response to analyses of past data, usually derived from historical trajectories of catch and effort or biological samples taken from past catches (Haddon 2001, Hilborn and Walters 1992, Walters 1996). Thus, planning for the future is based on perspectives of the past, often with a qualitative rather than a quantitative evaluation of the relationship between the two. Consequences of such retrospective assessments and speculative planning typically include overcapitalisation of fishing fleets relative to sustainable harvests, recognition of the need for regulatory adjustments only after over-harvest has occurred and decision making constrained by the financial and social hardships either being experienced or likely to be experienced as a result of management actions required to rebuild over-exploited resources. Difficulties then arise in reconciling the social, economic, stock, conservation, individual and institutional objectives of different stakeholders in a fishery.

Several authors have argued that we should be seeking alternatives to existing singular management strategies that attempt to maximise the prospects of meeting management objectives and instead designing management actions to maximise the information that can be derived from the responses of the system to those actions (Walters and Holling 1990, Walters 1986, Hilborn and Walters 1992). Ideally, such an approach (active adaptive management) would involve selecting alternative management strategies in the same way that different treatments are selected in designed experiments to highlight the contrasts between alternative hypotheses about the system. In reality, such an approach is difficult to implement, unlikely to be supported politically and often unfeasible given the logistics of most fisheries and their history (McAllister and Peterman 1992).

Use of simulations of the fishery system, however, obviate most of these issues and facilitate quantitative prospective exploration of management strategies (De la Mare 1996, Francis 1992, Francis and Shotton 1997, Smith 1994, Smith *et al.* 1993). Termed Management Strategy Evaluation (MSE, Smith 1993, 1994), the approach varies from more conventional approaches to fisheries stock assessments in several ways. First, MSE is focused on evaluating the medium to long term performance of management strategies, rather than on short term assessments of regulatory practice. Second, MSE is comparative rather than prescriptive, seeking to compare likely outcomes of a range of management scenarios rather than to prescribe actions that should be taken under an existing regulatory framework. Third, MSE seeks to compare the future status of a range of performance indicators with their desired values (objectives) under a range of management strategies rather than identify the objective (usually the level of fishing mortality that should be allowed) that should be pursued by an existing strategy. Fourth, MSE seeks to compare the performance of a range of candidate management strategies in terms of multiple performance indicators that reflect a diversity of stakeholder objectives, including social, economic and biological, instead of considering only a single performance indicator (usually the size of the harvested resource), that would reflect on only a single objective. Finally, MSE seeks to provide a system for comparing the performance of alternative management strategies against different stakeholders' objectives based on a common currency across all or most objectives.

Management Strategy Evaluations require a number of key elements for successful application. First, it is necessary to have the capacity to simulate the dynamics of the harvested stock or stocks (i.e., a fully parameterised a population dynamics model), the dynamics of harvest of the stock (a harvest model) and the interactions between the population and harvest models (collectively, an operating model). The operating model must be a credible representation of the actual biological and fishery system, necessitating considerable amounts of biological information and other data. Second, MSE requires the

identification of quantitative objectives, performance indicators and (preferably) target values for those performance indicators (Smith 1994). The management objectives must adequately reflect the desires of the stakeholders in the fishery if the results of MSE are to be accepted and influential (Stephenson and Lane 1995). Finally, specification of a range of alternative management strategies or scenarios by which the objectives might be achieved is required, together with the capacity to implement representations of those strategies in the simulation framework. These management strategies compared must be both feasible and likely to be supported. Thus, MSE requires not only a credible research base but also the active engagement of a diversity of stakeholders in the formulation of management objectives and strategies.

The GBR has many features that would provide for active adaptive management in practice, but the fact that it is a World Heritage Area and hence cloaked in political and environmental sensitivities means that management approaches are likely to be conservative and incremental rather than exploratory and adaptive. There are, however, different management strategies in place that directly or indirectly regulate the reef line fishery, although their relative or absolute performance has not been evaluated. The existence of several agendas for managing use of the GBR means that evaluation of these different strategies is complicated and often undertaken from the perspective of apparently competing objectives. Management Strategy Evaluation has considerable potential benefit in such an environment.

We sought to develop a Management Strategy Evaluation system for application to the Reef Line Fishery operating in the GBR Marine Park and World Heritage Area. We have developed a detailed Effects of Line Fishery Simulator (ELFSim) based on prior, more rudimentary models of coral trout populations on the GBR (Campbell *et al.* 2001, Mapstone *et al.* 1996e, Walters and Sainsbury 1990). In developing credible population dynamics and harvest models, we sought also to engage the major stakeholders in the fishery and the World Heritage Area in the research to 'anchor' the Management Strategy Evaluations to the objectives, expectations and management directions of people involved in the system. What follows is a summary these efforts and the results from them.

Methods

We developed an operating model for the harvest of common coral trout (*P. leopardus*) over the highly fragmented coral reef habitats of the Great Barrier Reef. The model has biological and harvest components representing the population dynamics and harvest of the resource by commercial, charter and recreational. The geographic scope of the model was restricted to the boundaries of the GBR Marine Park (Fig. 1). A general description of the operating model is presented below, whilst the details of the model are described in Appendix B.

The Biological Component of the Operating Model

The biological component of the operating model simulates the population dynamics of a species subject to harvest, in this case *P. leopardus*. The population dynamics model in ELFSim was developed initially for common coral trout because a) it is the primary target species in the RLF, especially for the commercial sector that accounts for the majority of catch, and b) it was the only target species for which there was sufficient biological information available to parameterise an operating model when the work commenced. This component of the operating model is an extension of the model first developed by Walters and Sainsbury (1990) and subsequently modified as described by Mapstone *et al.* (1996e) and Campbell *et al.* (2001). In refining the model for this work, we sought to provide a model that could be extended to other species with generally similar demographics and life-history strategies as sufficient data became available. The model currently is being developed further for application red throat emperor, *Lethrinus miniatus*.

The coral trout resource on the GBR is represented as a meta-population of post-settlement individuals in which the sub-populations interact only via larval dispersal. The model allows for movement of larvae among the sub-populations but movement of animals aged 1 year and older is precluded, because there is little evidence for movement of such animals (Davies 1995a,b). Each post-settlement sub-population is associated with a single reef.

There are currently 3822 sub-populations depicted in the model. This number is 1182 greater than the gazetted and mapped number of reefs in the GBRMP because we have included 'virtual reefs' in the operating model. Virtual reefs were included because there were 1182 sites¹ in the catch and effort data for the commercial or charter fishery from which catches of coral trout were reported but where there was no mapped coral reef. Two parsimonious options were considered for handling these records: a) assume that they were erroneous and disregard them; or b) assume that they were legitimate and reflected fishing on reef substratum that was not mapped and include them in the simulations. We chose the latter because we were aware of numerous such uncharted, usually submerged reefs and shoals in the GBR and commentary from fishers was that such reefs are targeted from time to time. Because post-settlement populations are associated with reef habitat, we had to assign measures of habitat (reef perimeter²) to these sites in order to sustain populations that could be harvested at them. Catch and effort from these sites generally were amongst the lowest of all records for sites where reef was mapped, accounting for only 3% of all reported effort during 1989-1996. We allocated virtual reefs to each 'reef-free' site with a reef perimeter equivalent to the reef with the most similar catch from the surrounding nearest 10 reefs, thus preserving the spatial patterns in attributes of reef perimeter, catch and effort.

The population dynamics model is age-, sex- and size-structured and assumes that the number of 0-year-olds is related to the size of the reproductive component of the population according to a Beverton-Holt stock-recruitment relationship with steepness of 0.5³ (Francis 1992). Several sources of process error (*sensu* Francis and Shotton 1997) such as variation

¹ Catch and effort data are reported against a set of geographic **grids** (30'x30') and, optionally, **sites** (6'x6') within grids and not by reef. In this report, 'site' refers to the 6'x6' cells in this reporting system. How we relate catch and effort data to reefs is described in Appendix B. Note also that there were a total of 2098 'reef-free' sites from which catch was reported, 1182 within the domain for this report.

² Perimeters of gazetted reefs were provided by the GBRMPA.

³ Sensitivity of the results to different values of steepness was explored when developing the model. The impact of changing steepness on modelled population dynamics is illustrated in Appendix C.

in natural mortality, larval survival and recruitment (to the post-settlement sub-populations) are included in the biological component of the operating model.

Growth in the length of an individual is described by the von Bertalanffy growth equation. Variation in growth among individuals is modelled by dividing each cohort into ten growth groups and assigning different von Bertalanffy parameters to each group. All animals within a growth group are assumed to grow according to the same growth curve. The values for the parameters determining growth in each group and the proportion of 0-year-olds in each growth group were determined by fitting a model to length-at-age data, after accounting for gear selectivity (Annex B.3, Appendix B). Mass of individuals is derived from the standard allometric equation relating weight to length, parameterised from field data (Table 5).

Natural mortality, fecundity, sex-change and growth are reef dependent in the model, thereby allowing for systematic as well as stochastic variation in these parameters at any scale of reefs or greater. Data to estimate such parameters, however, were available for only a small subset of reefs and so variation in these parameters was set to be spatially uniform over all reefs in the simulations reported here. Temporal variation enters the model in both the average and variance of age-specific natural mortality, facilitating the simulation of catastrophic events and time-trends in natural mortality as well as stochastic processes. It is assumed that catastrophic events have the same impact across all of the reefs included in a given run of the model, thus providing a framework within which to explore some of the possible impacts of large-scale or global environmental events on the dynamics of, and the fishery for, coral trout. Natural mortality is age-specific and assumed to be greater for younger animals (ages 0 and 1, Mapstone *et al.* 1996e, Campbell *et al.* 2001) than for older animals, with fish aged 2 and older having a constant rate of natural mortality (M) of 0.3 yr^{-1} (Table 5). The values implemented for natural mortality-at-age were derived by reference to previously published estimates and based on the assumption that natural mortality will decline with size (and age) and with reference to the observed age-structure of coral trout on closed reefs (Mapstone *et al.* 1996e, Campbell *et al.* 2001).

Recruitment to each reef-associated sub-population occurs at the start of each year (Ferriera and Russ 1994, Russ *et al.* 1996) and all fish are assumed to recruit as females (Adams *et al.* 2001, Adams 2002). The number of 0-year-olds on each reef at the start of each year comprises contributions from spawning on that and other reefs (Mapstone *et al.* 1996). The relative contributions of larvae to each reef from itself ('self seeding') and from other reefs is specified by a larval dispersal matrix that defines the fraction of larvae that move from each reef to each other reef. Three alternative approaches for deriving the larval dispersal matrix are included in the model: i) a uniform distribution of larvae, where the probability of larval exchange between reefs is distance-independent; ii) a distance-based distribution of larvae, in which the rate of larval seeding between reefs declines exponentially with increasing distance between them; and iii) a pre-specified larval dispersal matrix derived by modelling likely dispersal scenarios from hydrodynamic models of the GBR (James *et al.* 2002).

A uniform distribution of larvae (alternative i) seems unrealistic for scenarios including large numbers of reefs over large distances and the larval mixing matrices derived from hydrodynamics (alternative iii) currently are available only for the Cairns section of the GBR Marine Park. Accordingly, the simulations reported here are based on the inverse-distance model of larval dispersal⁴. Use of larval dispersal rates determined from models of larval advection and larval behaviour are clearly more desirable and the models to provide such inputs currently are being extended to cover the entire GBR.

⁴ Comparisons of results from using the distance-based and hydrodynamic model-based larval mixing matrices for the Cairns Section of the GBRMP showed that the methods produced similar results when aggregated over all reefs, but increasingly different results as smaller scales were compared.

Table 5: Default values of the key parameters of the operating model used when assessing the performance of alternative management strategies.

Parameter	Base-case value	Source
Maximum age	18 yr	Mapstone <i>et al.</i> 1996, Ferreira and Russ 1994.
Mortality @ Age 0	0.4 yr ⁻¹	Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001.
Mortality @ Age 1	0.35 yr ⁻¹	B. Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001
Mortality @ Age 2+	0.3 yr ⁻¹	B. Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001
Temporal variation in natural mortality	0.05	Mapstone <i>et al.</i> 1996
Temporal auto-correlation in natural mortality	0	Assumed
Variation in 0-year-old survival	0.6	Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001.
Spatial correlation in 0-year-old survival	0.5	Assumed
Larval self seeding	0.1	Jones <i>et al.</i> 1999, James <i>et al.</i> 2002.
Larval retention probability	0.05	Mapstone <i>et al.</i> 1996
Larval dispersal matrix	Inverse distance	Pre-specified [*]
Time of larval settlement	January	P. J. Doherty pers. com.
Steepness, h	0.5	Pre-specified [*]
Length-mass parameters, $\lambda_1(b_1)$, b_2	-11.03, 2.97	G.R. Russ pers. com, Mapstone unpub. data.
Length-at-50%-selectivity (FL)	322 mm TL	Fulton 1996, Fulton <i>et al.</i> 1999
Length-at-95%-selectivity (FL)	375 mm TL	Fulton 1996, Fulton <i>et al.</i> 1999
Length-at-50%-maturity (FL)	280 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.
Length-at-95%-maturity (FL)	360 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.
Length-at-50%-sex change	450 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.
Length-at-95%-sex change	500 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.
Extent of density-dependence in catchability	0	Assumed
Variability in effort-fishing mortality relationship	0.3	Assumed
Closure infringement parameter, M	0.999	Pre-specified [*]
Habitat scalar, h_s	0.25	Pre-specified [*]
Effort concentration parameter	0	Pre-specified
Time step	1 month	Pre-specified

* Alternative values for these parameters were explored to assess the sensitivity of the model results to changes in their value. Results of these sensitivity analyses are summarised in Appendix C.

Sex change is implemented by modelling the fraction of the animals that are mature and those that are male by separate logistic functions of length, thereby allowing for the known variation in both the onset of maturity and size of sex change (Adams 2002).

The number of 1-year-olds in each growth group on each reef in each year is determined from the number of zero-year-olds on the reef in that growth group in the previous year modified by the density-dependent mortality between ages 0 and 1 plus the impact of random environmental variability and 'recruitment pulses'. This means that 'recruitment pulses' can lead to higher or lower numbers of individuals surviving from age 0 to age 1. Recruitment pulses can be tuned to be effective over different scales, so that groups of reefs can receive recruitment pulses at the same time. The centres for the 'recruitment' pulses in the current simulations were distributed randomly over the Great Barrier Reef.

The Harvest Component of the Operating Model

The harvest component of the operating model (or 'effort allocation model') is required to simulate the harvest of the population(s) represented by the biological population dynamics model (described above). A harvest model should represent the distribution of fishing effort over the fishing ground and simulate shifts in fishing effort (in space and time) in response to imposed management strategies. The harvest model also must impose fishing mortality on the resource and, optionally, keep track of resulting catch.

The harvest component of the operating model in ELFSim allows for multiple effort-classes, representing different fishing fleets. Each effort-class can be divided into sub-groups that operate within a pre-specified spatial domain (or 'region'). Regions are currently implemented as those defined by Mapstone *et al.* (1996a) based on the operational characteristics of the commercial fleet during the early 1990s. Regions need not be mutually-exclusive, and we have implemented an additional 'region' defined as the entire GBR Marine Park to cater for vessels that roam widely and do not restrict their operations to particular regions within the GBR. The work reported here is based on three effort classes, parameterised to represent the commercial, charter and recreational sectors of the reef line fishery, each operating over the entire GBR Marine Park.

The effort allocation model is not intended to mimic the decision making behaviours of individual skippers but simply to capture the net effect of all such individual decisions when aggregated up to fleet level. Whilst models of individual-based (Mapstone *et al.* 1999) or group-based (Little *et al.* 2001) vessel dynamics are being developed, they were not available for the current sets of simulations. The effort allocation algorithm used, however, provided adequate representation of the historical patterns in fishing over the GBR given the constraints of data available to parameterise the models and the run-time necessary to complete simulations.

From 1965 to 1998, the effort and catch allocated to each site, and thence to reefs within sites, is the actual (reported) catch of coral trout and the associated effort or values interpolated backwards from those reported data. Actual catch and effort data exist from 1989 onwards for the commercial sector, from 1996 onwards for the charter sector and bi-annually from 1998 for the recreational sector. Therefore, for the years prior to these dates, catches are interpolated linearly for each calendar month backwards to a value of zero in 1965. The starting points for these interpolations are the years immediately prior to the earliest year of actual data for each sector, when the catch for each month is assigned the average of reported values in that month from 1989-92 for the commercial sector, 1996-1998 for the charter sector and in 1998 for the recreational sector. This strategy preserves in the historical dynamics the seasonality observed in the fishery since 1989.

The spatial distributions of catch and effort data for the commercial and charter fleets were as reported in the respective logbooks, since in both cases skippers have to report their fishing location daily. Few analogous data exist for independent recreational fishers, however, so we inferred the distribution of recreational fishing based on boat ramp survey data (Blamey and Hundloe 1993, Higgs 1996, Mapstone *et al.* 1997, Higgs, Mapstone unpub. data) and recreational fishing diary data (Higgs, Mapstone, QFS unpub. data).

Recreational fishing effort (and catch) is modelled to be distributed in proportion to the inverse of the distance of the centroid of each site from major known access points along the coast (typically boat ramps), with a limit of 60nm, beyond which we considered recreational fishers would not normally travel. The total amount of catch of coral trout and fishing effort available for distribution from each access point was based on data collected for each coastal statistical division in 1998 by the QFS during bi-annual surveys of recreational fishing in Queensland. This procedure captured the major regional patterns in recreational fishing activity and meant that most recreational fishing was deemed to occur in near-shore sites, with negligible amounts of effort being assigned to reefs well off-shore. This is consistent with previous research results (Blamey and Hundloe 1993, Mapstone *et al.* 1997, Higgs 1996).

The effort allocation algorithm allocates effort dynamically over the spatial domain at each time step (in this case, 1 month) during the projection period (1999-2025). The basic information required about fishing effort for the projection period is the annual level of effort (by effort-class and region) for the first year of the projection period (1999) and either the rate at which annual effort increases or the specified proportion of the initial (1999) effort to be extant after some period of transition. For example, effort may be set to increase annually until it reaches 150% of the initial effort after five years. This last option is used to avoid severe discontinuities in effort between the historical and projection periods by allowing a gradual change in total effort over time. Such changes in effort are assumed to be linear.

The total annual effort is divided among the time-steps within the year according to intra-annual patterns known from recorded data, to preserve seasonality in the distribution of effort during the projection period,. This is accomplished by selecting a year at random from the period for which there are real data (1989–98 commercial, 1996–1998 charter, and 1998 recreational), calculating the fraction of that year's effort that occurred in each time step during the year, and using these fractions to distribute the total effort allocated for the future year among the time steps within that future year.

Once the effort available for allocation in each time-step and each effort-class is determined, four steps are involved in the spatial allocation process (in this case to 6'x6' sites⁵). These steps make use of the catch rates and effort levels for each effort-class in each site in the past (during both initialisation and projection periods) and the management status of each site or reef (open or closed to fishing – see below). The four steps are:

- a) Sites that were fished in the previous time step are ranked according to their weighted CPUE up to that time. The weighted historical CPUE is calculated as the ratio of summed previous catches to summed previous effort, where the summed previous catches and effort is multiplied at each time step by a discount factor that effectively down-weights the influence of older data. Changing the discount factor changes the period over which catch and effort from a given step influences future decisions. Historical summations of catch and effort can be set to sum over all preceding months in the same calendar year or sum over the current calendar month (*e.g.*, January) in all previous years. The latter strategy has been used in the simulations presented here because of the known seasonality in the dynamics of the fleets in the RLF.
- b) Effort is first allocated to the highest ranked (most desirable) site and then consecutively to the next ranked sites in turn, until there is no effort left or all of the sites that were fished in the previous time step have been assigned effort. The effort allocated to each site is the weighted average historical effort for that site in a given month multiplied by a 'concentration factor' that determines the degree to which effort is likely to aggregate at more 'desirable' (in terms of CPUE) sites at the expense of lower ranked sites. The effort concentration parameter was set to

⁵ The operating model has since been refined to allocate effort directly among reefs based on an *a priori* disaggregation of the reported effort and catch data from the grid-site reporting frame to reefs, as described above. The model then has only the reef specific data available, thus obviating the need to translate between grid-site and reef frames of reference at each iteration of the effort allocation – harvest process.

0 for simulations reported here, effectively meaning that future distributions of effort were dictated principally by past distributions without any additional favouring of sites with higher CPUE.

- c) If effort remains to be allocated after step b), it is allocated to the sites that were not fished in the previous time step but had been fished at some earlier time. As in steps a)-b), these previously fished sites are ranked according to CPUE and effort is allocated to them in order until either there is no effort left, or all of these sites have been assigned some effort.
- d) If unallocated effort still remains, it is allocated randomly to sites in small (5%) portions until all remaining effort is allocated. This step allows for exploration of new fishing grounds and is the only way that effort can be allocated to sites that have not been fished historically.

The proportion of annual effort allocated to “virtual” reefs is constrained to fall within the proportions reported in the logbook data from 1989 to 1998 because of the uncertainty about how much habitat really exists in such sites and hence uncertainty about the standing of CPUE from those sites relative to others.

All sites in the domain for which a simulation is run have a ‘management status’ of open or closed to fishing. Management status is set as a variable (M), however, valued between 0 (open) and 1 (completely closed). Allowing for values of M between 0 and 1 facilitates setting some level of infringement of putative closures. The effective effort assigned to a site is calculated as that effort that would have been allocated to the site if it was open to fishing multiplied by $1-M$. The value of M can vary with time, allowing for seasonal (intra-annual) or rotational (inter-annual) closures to be implemented.

Closures were implemented by site (6' x 6' map reference) and the closure status of each reef within a site was determined by proportion of its perimeter that lay within that site. If 50% or more of a reef's perimeter was within a closed site (or sites), that reef was deemed closed to fishing, otherwise it was considered open for the purposes of effort allocation. Where a reef straddled a boundary between a closed site and an adjacent open site, the potential existed for that reef to be allocated some effort from the open site and thus be subject to fishing mortality. This meant that there was some ‘leakage’ of fishing mortality onto closed reefs, effectively mimicking some unspecified level of infringement. The consequence of this phenomenon, as with infringement in reality, would be to reduce the effectiveness of closures and diminish contrasts between management scenarios involving different levels of closure. Because of this issue, we set a very low level ($\ll 1\%$) of ‘deliberate’ infringement (as described above) in the current simulations. This uncontrolled infringement has been removed since obviating the use of grid-site geo-referencing in ELFSim (the Effects of Line Fishing Simulator) by prior allocation of recorded effort to reefs. The implications of various known rates of infringement under alternative closure regimes is being explored elsewhere (Little *et al.* in prep.).

Provision also exists for both spatial (edge effects) and temporal patterns in infringement of closed areas. The edge effect specifies that effort inside a closed area diminishes exponentially with distance from the closure border. The parameters of the decay curve can be varied to change the effective ‘hardness’ of closed area boundaries. We used a value that meant that infringement effort would be at 50% of its boundary value 1 site (~6 nm) into a closed area. Temporal patterns in infringement are modelled in similar fashion, with exponential changes in infringement from a given time (such as the time of closure) allowing for decreasing or increasing infringement over time.

Catches

The catch (in mass) of fish from each reef in each time-step (month) is calculated separately for each effort-class represented in the operating model. The catch is a function of the biomass available to the fishery on each reef at the start of the month, the amount of effort from each sector of the fishery applied to the reef in that month, and sector-specific selectivity and catchability functions. Selectivity for the commercial fishery was estimated by Fulton (1996, Fulton *et al.* 2000) and currently is applied to all effort-classes in the model

because there is no empirical basis for estimating selectivity for the charter and recreational sectors. The harvest model distinguishes between retained and discarded catch, according to the Minimum Legal Size Limit (MLS) set for the fishery. Harvest (retained catch) is all that catch which is greater than the MLS (Appendix B). Illegal retention of under sized catch and post-release mortality is allowed for by designating a proportion of the under sized catch that dies as a result of capture. We assumed that 15% of under sized catch died in the simulations reported here.

Catchability coefficients for each effort-class are estimated by reef from reported catch and effort data (Appendix B). Since this approach cannot be applied to reefs for which there are no catch and effort data, catchability coefficients for such reefs must be assigned rather than estimated. The commercial catchability coefficient for a reef without catch and effort data is taken to be that from the nearest reef for which catch and effort data exist. Catchability coefficients for such reefs for the charter and recreational sectors, however, were set to zero because applying a similar approach to that used for the commercial sector resulted in an unrealistically uniform spread of charter and recreational effort over the entire GBR. This problem arose largely because of the relatively small number of reefs from which catch and effort data either existed or could be inferred for the charter and recreational sectors. This approach, however, prevented recreational effort occurring in the future on reefs that had not been visited by recreational fishers in the past.

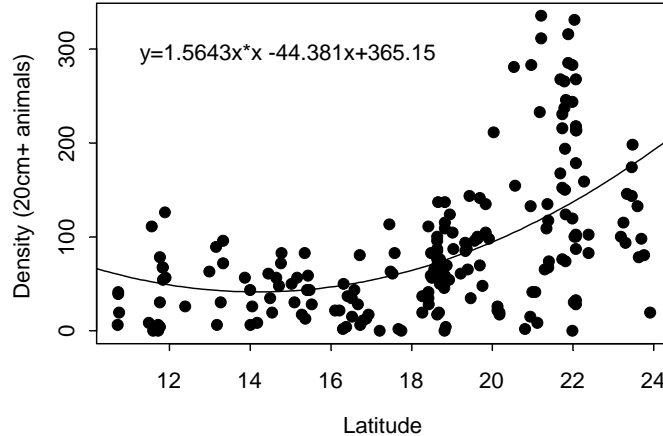
Initialisation

For the purposes of the simulations, we assumed that each reef-associated sub-population was at pre-exploitation equilibrium with the corresponding age- and sex-structure at the start of 1965. This starting date was arbitrary and not related to any assessment of the actual history of the fishery. It should be noted, however, that the choice of date would not impact on performance of the operating model, or projections from it, given that the only fishery data used in tuning the model were from 1989 to 1998.

The pre-exploitation sub-population size for each reef is calculated as a function of reef perimeter and population density data for common coral trout from reefs closed to fishing. A 'Habitat Scalar' parameter is used to convert the product of (predicted) population density and reef perimeter to the numbers of animals in the unfished sub-populations. The Habitat Scalar provides a means of setting virgin population sizes such that after running the operating model from 1965 to 1989, the actual catches observed during the 1990s are realised in the simulation data. The value of the 'Habitat Scalar' is largely arbitrary, however, because we have a poor understanding of the amount of reef in the GBR habitable to coral trout or its relationship with mapped reef perimeters. Thus, observed catches could be realised with relatively low fishing mortality and a large value for the 'Habitat Scalar', or higher fishing mortality and lower values for the Habitat Scalar. We chose a value for the Habitat Scalar that minimised the frequency of reef-level extinctions of coral trout in the simulations (see below) and led to levels of depletions of reef-specific sub-populations that approximated what would be expected given the available size and age data from heavily fished reefs. We also ran simulations with different values of the Habitat Scalar to assess the sensitivity of the outcomes to changes in its value.

Regional variation in pre-exploitation equilibria ('carrying capacity') of reefs is derived from past underwater visual surveys that established a relationship between the population density of fish larger than 20cm (TL) and latitude (Ayling and Ayling 1986, 1992a) (Fig. 60). This regional variation is captured by scaling each reef to the value corresponding to its latitude in the curve in Figure 60 and dividing by the value at the minimum of the curve (16.5°S - *i.e.* the scalar is 1 for reefs at 16.5°S). This scalar can be modified to realise different distributions of the resource at the start of projections, if desired.

Figure 60: Relative population density of common coral trout (*P. leopardus*) within the GBR Marine Park estimated from underwater visual surveys between 1983 and 1995 plotted against latitude.



The age- and size-structure of each reef sub-population at the start of the first year when a management strategy is to be applied (*i.e.*, the first year of projections, 1999) is determined by projecting the population from pre-exploitation equilibrium (in 1965) to the start of 1999 with random variation in recruitment and natural mortality and subject to realising the reported catches prior to 1999. The allocation of historical catches to reefs in this initialization, however, can result in population extinctions on individual reefs prior to the projection period. If extinction occurs on a reef, the initialisation process is repeated after incrementing in 5% steps the Habitat Scalar for that reef until no extinctions occur. Effectively, this means tuning the population model to realise the reported catches prior to the projection period, given the distribution of the resource and fishing effort, under the assumption that local extinctions are unlikely to have occurred in the past given contemporary regulations and levels of fishing. This seems a reasonable proposition, given dispersal of larvae among reefs and that the Minimum Legal Size of harvest of common coral trout (38cm Total Length) is a size by which all fish would have reproduced in at least one year. Effects of changing the value of the Habitat Scalar are shown in Appendix C.

Projections and Management Scenarios

Management scenarios are implemented during the projection period by varying annual effort, access to areas for fishing and parameters determining the interaction between the biological and harvest models, such as selectivity and minimum legal size limits. These constraints can be applied at the beginning of the projections or as time varying measures, but are always pre-specified. That is, there is no dynamic feedback between harvest strategies and stock dynamics. Thus, only static strategies are considered in this report.

Evaluations proceed by running the operating model from 1965 to the end of the period for which real data are supplied (the initialisation period) and then introducing the desired changes in parameters that define specific management scenarios for evaluation. Random processes in the population dynamics mean that each initialisation will generate different starting conditions for projections. The model then runs for a defined projection period (in this case 27 years), usually defined by the time taken for key variables to stabilize.

Repeating runs with the same management scenarios allows assessment of the impact of variation in population dynamics and effort allocations on the results for that management scenario. Running the same scenario with different parameters for the underlying operating model allows assessment of the robustness of the results to uncertainties or errors in the model assumptions. A wide range of reef-specific data can be collected at each time step, including catch and effort from each effort-class, available biomass, spawning biomass, fishing mortality and size and age measures for the population and the catch.

ELFSim is designed with a high degree of modularity, such that alternative forms of core components can be developed independently and 'plugged in' to replace or compare with

existing components. Thus, alternative procedures for modelling effort allocation via explicit representation of vessel behaviours will be incorporated with minimal need to modify the population dynamics in the operating model. Much of the input data with which the model is parameterised is provided via free-standing MS Access databases or GIS shape files, allowing straightforward re-parameterisation of the model for new species, fleets, or spatial domains. Similarly, output is written to MS Access databases for analyses with other software. ELFSim also contains a graphical user interface (GUI) from which many of the run-time choices of parameters can be set and summary results can be viewed as reef-specific statistics and plots. The GUI aids with running the software and is particularly useful for discussing the software and the results it generates with stakeholders.

Setting Objectives, Performance Indicators and Management Strategies

ELFSim has the potential to run a great diversity of scenarios to investigate the impacts of changes in very many parameters governing both the population dynamics of the harvested species, their harvest and the strategies by which they are managed. The primary purpose of the tool, however, is to explore the performance of key performance-indicating variables under a range of selected conditions and to allow systematic comparison of the merits of alternative strategies for regulating harvest. The choice of the management strategies to be compared, the desired outcomes (objectives) against which their success will be measured, and the performance indicators by which they are evaluated is central to the MSE process.

We sought input from a wide range of stakeholders in the RLF to identify key objectives for the fishery and the feasible management strategies by which those objectives might be pursued. We did so via formal and informal workshops and meetings over a period of two years. Our objective in this process was not to seek consensus among the different interest groups, but rather to identify the range of specific, quantifiable objectives and strategies that were considered desirable or feasible by one or more groups. These objectives were subject only to the constraint of what was amenable to evaluation using ELFSim. It is important to note that during this process several other objectives and management strategies were identified that were important, but could not be evaluated with ELFSim.

Formal workshops with a diversity of stakeholders were convened in December 1999 and November 2000. The workshops were convened in conjunction with meetings of the Reef Line Fishery Management Advisory Committee (ReefMAC) to align the MSE process with policy development and management process for the fishery. In the first workshop, we presented the concept of MSE and the modelling framework being developed, demonstrated ELFSim and initiated discussion about what was required in objectives and strategies, and what sorts of performance indicators might be appropriate.

Following that workshop, we met informally and separately with a range of people from different stakeholder interests to clarify their interests and assist in the formulation of the specific objectives most important to them. During these meetings, we followed the hierarchical process suggested by Chesson *et al.* (1996) for resolving objectives and their associated performance indicators. The groups we worked with included: management agencies (QFS, GBRMPA); enforcement staff (QBFP); commercial, charter and recreational fishers and their peak representative bodies; fish buyers; and members of conservation non-government organizations (CNGOs), including the North Queensland Conservation Council, WWF, the Queensland Conservation Council, Australian Marine Conservation Society, and the Marine and Coastal Community Network.

The second formal workshop was focused on selecting a set of management scenarios for application in ELFSim and determining specific, quantitative objectives and performance indicators by which to compare their performance. Summary reports from the first and second formal workshops are provided in Appendices D and E respectively.

Two main management approaches were recommended by stakeholders for evaluation with ELFSim: effort regulations and area closures. These approaches respectively represented the main instruments being used at the time by the QFS and GBRMPA to manage the RLF and conserve biodiversity on the GBR. We were asked to consider four levels of future

fishing effort and three area closure regimes in combination, providing a total of 12 strategy sets. The effort strategies comprised projections with effective fishing effort for each of the recreational, charter and commercial sectors set at the then-current (1999) levels, those reported in 1996, half that level and 1.5 times that level. Each of these effort levels was to be assessed under the current GBRMP area closure regime (~16% by number and reef perimeter, 23% by reef area) and two higher levels of closure (~30% and ~50% by reef perimeter). On examination, we found the then current effort levels to be similar to those in 1996, and hence redundant, thus reducing the number of strategy sets from 12 to 9.

The rationale for the chosen effort levels was that they bracketed the range of effort regulation options considered possible and feasible at the time. The then draft management plan for the coral reef fin fish fishery had pinned a major review event to effort levels exceeding those present in 1996 for the commercial sector or increases of more than 5% per year in the recreational and charter sectors. Accordingly, the approximate level of effort in the RLF in 1996 was considered a 'status quo' reference point against which to assess the effects of either allowing effort to grow or reducing effort in future. It was considered by stakeholders at the workshop that effort was unlikely to grow beyond 150% of the 1996 reference level and that effort reductions beyond 50% of the 1996 level were unlikely to be either necessary or feasible unless there were dramatic changes in perceptions of current stock status. Thus, expectations of future levels of effort were between these extremes.

In considering these effort levels, we infer that our projections have the *effective* level of effort equated to multiples of that in 1996. We did not attempt to model 'effort creep', there being no data from which to parameterise changes in effectiveness of a unit of effort. Thus, our projections are appropriate either for the designated level of future effort with static effectiveness (*i.e.*, with no 'effort creep') or for some discounted level of effort with increased effectiveness such that the net effectiveness of the effort is equivalent that modelled.

The three regimes of area closures were chosen following extensive discussion of the expected increase in the amount of coral reef habitat closed to fishing under the GBRMPA's Representative Areas Program, then in the early stages of development. Whilst the principles under which the Representative Areas Program was being developed were well known, the results of the program in terms of amounts of area gazetted as 'no take' were not known. It was agreed, however, that there would not be less reef habitat closed under the program than closed currently. Accordingly, two scenarios of increased closure were chosen that represented significant increases over current ('status-quo') closures, up to what was considered to be the most extreme level of closure likely (50% by reef perimeter).

A single additional management instrument also was suggested for evaluation. This involved reducing the minimum legal size limit (MLS) for harvest of coral trout from the current 38cm Total Length (TL) to 30cm. This would represent changing the MLS from a length by which most fish would have reproduced in at least one year, to the size at which only 50% of individuals would have matured (Adams 1996, 2002, Adams *et al.* 2001). This strategy was to be applied under the otherwise 'status quo' conditions of current closures and current (=1996) effort⁶.

In addition to the above strategies, we ran strategy sets of zero closures for each level of effort (ie, entire fishing ground available) and a zero effort regime (fishery completely closed) to provide 'base case' reference sets for each of the above strategies. The latter strategies were not proposed as either feasible or desirable by stakeholders, but used by us principally in the process of verifying the behaviour of the models. The strategy sets evaluated for this report are summarised in table 6.

Objectives

The primary qualitative objectives of importance to various stakeholders related to:

- a) the protection of some parts of the GBR from all impacts of line fishing;

⁶ An error in setting up the runs in ELFSim meant that the reduced MLS limit strategy was actually run under the scenario of 50% closure and current (1996) effort, meaning that the appropriate control set against which this option was compared involved 50% closures, 1996 effort and current MLS.

- b) conservation of coral trout stocks, specifically their reproductive capacity, genetic diversity and age and the size structures of unfished populations;
- c) maintenance of satisfactory levels of stock available for harvest (above the MLS);
- d) provision for economic viability of the commercial and charter sectors; and
- e) provision of satisfying fishing experiences for recreational fishers.

Some of these qualitative objectives, such as maintaining genetic diversity, could not be addressed directly or even indirectly by ELFSim. Other objectives, such as maintaining economic viability and fishing satisfaction, could not be addressed directly but could be translated into specific objectives which were considered necessary or sufficient conditions for meeting the more general qualitative objectives. Accordingly, the following specific objectives for the RLF were agreed during the second formal workshop.

Conservation: Spawning biomass in areas closed to fishing should remain above 80% of virgin spawning biomass for 90% or more of the time, and preferably close to 90% of virgin spawning biomass. This was an accepted surrogate for the objectives of protecting portions of the coral trout stock from fishing and conserving populations in closed areas since it was expected that the conservation objective would only be met if the (protected) populations were in most respects (size and age structure, sex ratio) close to their unfished state. We report also on the status of 'available' biomass in the closed areas and the status of spawning biomass in areas open to fishing as additional indicators of the conservation status of coral trout stocks.

Status of Harvestable Stock: Available biomass (fish above the MLS and available to the gear) in areas open to fishing should remain above 30% of virgin levels 90% of the time. The limit reference value of 30% was arbitrary but considered to be the minimum level to which available biomass should be allowed to fall, assuming reasonable protection of populations in closed areas. We report also on the likelihood that available biomass would exceed higher reference points, specifically 50% and 70% of virgin available biomass.

Economic Viability: Commercial catch rate was accepted as an indicator of economic viability for the commercial fishery. This assessment was made in the absence of explicit economic models of the RLF and under the assumption that the price of product would change roughly in proportion to costs of harvest, or more advantageously, and so net value of revenue to operators would be dependent mainly on the rate at which fish could be harvested. Fishers cited the early to mid 1990's as a relatively profitable and stable period in the fishery and suggested that catch rates close to those from 1993-96 would ensure economic viability of most commercial line fishing operations. Accordingly, the objective became that annual average CPUE for the commercial sector of the RLF should remain above 80% of the 1993-96 average 90% of the time. For consistency, we also report performance against analogous objectives for the charter and recreational sectors, with reference values of the average of 1996-98 and 1998 data respectively⁷.

⁷ The specific target CPUE's for the charter sector from the workshop were absolute quantities of 1 and 4 coral trout per angler per day for day-trip and extended operators respectively. Since these were estimated as the catch rates from survey data, and for comparison with objectives for the commercial sector, we translated these reference points to be relative to the data from which they were estimated.

Table 6: Stakeholder specified objectives, reference points and associated performance indicators (PI) for the future of the Reef Line Fishery in the Great Barrier Reef Marine Park by which the performance of preferred Stakeholder Management Strategy Sets were compared. MLS is the Minimum Legal Size limit used for harvest of catch (38cm TL \approx 36cm FL; 30cm TL \approx 28cm FL). Commercial, charter and recreational effort in the GBR Region in 1996 were estimated as 74,543, 25,317 and 165,923 line days respectively. Reference Strategy Sets were added by us for model verification and as points of reference to illustrate some results from Stakeholder Management Strategy Sets.

OBJECTIVES			
Objective	PI	Reference Point	
Maintain spawning biomass (SB) on closed reefs	SB _t : Virgin SB	P(SB : Virgin SB > 0.8) > 0.9.	
Maintain available biomass (AB) on open reefs	AB _t : Virgin AB	P(AB : Virgin AB > 0.3) > 0.9.	
Maintain economically viable commercial CPUE	CPUE _t : CPUE ₁₉₉₃₋₉₆	P(CPUE : CPUE ₁₉₉₃₋₉₆ > 0.8) > 0.9.	
Maintain acceptable charter fishing CPUE	CPUE _t : CPUE ₁₉₉₆₋₉₈	P(CPUE : CPUE ₁₉₉₆₋₉₈ > 0.8) > 0.9.	
Maintain acceptable recreational catch CPUE	CPUE _t : CPUE ₁₉₉₈	P(CPUE : CPUE ₁₉₉₈ > 0.8) > 0.9.	
Provide reasonable chance of catching big fish	Prop ⁿ Catch >50cm	Not specified	
Maximise commercial harvest, given other objectives	Total Com. Harvest	Not Specified	
Minimise year-to-year variation in catch	(C _v -C _{v-1})/Avg Catch	Not specified	
STRATEGIES			
Stakeholder Management Strategy Set	% Closure	Effort	MLS (cm TL)
Current closure, Low effort	16	0.5 x 1996	38
Status Quo	16	1.0 x 1996	38
Current closure, High effort	16	1.5 x 1996	38
Medium closure, Low effort	30	0.5 x 1996	38
Medium closure, Current effort	30	1.0 x 1996	38
Medium closure, High effort	30	1.5 x 1996	38
High closure, Low effort	50	0.5 x 1996	38
High closure, Current effort	50	1.0 x 1996	38
High closure, High effort	50	1.5 x 1996	38
Reduced Minimum Legal Size	50	1.0 x 1996	30
Reference Strategy Sets			
Zero closure, Low effort	0	0.5 x 1996	38
Zero closure, Current effort	0	1.0 x 1996	38
Zero closure, High effort	0	1.5 x 1996	38
Zero Closure Zero Effort	0	0	N/A

Fishing Satisfaction: Fishing amenity or ‘satisfaction’, rather than the quantity of catch, was a key concern for recreational and charter fishers, especially since both sectors were subject to possession limits and the charter sector’s economic viability rested on being able to provide valued recreational experiences to clients. Whilst satisfaction entailed many features that were not amenable to consideration using ELFSim, having a good chance of catching a ‘good size’ (trophy) fish was considered an important component of satisfaction for recreational anglers. Whilst a definition of a ‘good sized’ coral trout (50cm TL) was agreed at the workshop, a specific objective for the frequency with which such fish should be caught could not be formulated and so we simply report the proportion of the catch that would exceed 50cm as an indicator for the qualitative objective of ‘having a reasonable prospect of catching a trophy fish’.

In addition to the above objectives, stakeholders stated two additional qualitative objectives without specific targets: that commercial harvest should be maximised and variation in catch should be minimised, subject to achieving the above objectives. Since no reference points were stated for these objectives, we simply report on the status of catch and inter-annual variation in catch under each strategy set. We also report two other variables of relevance to the proposed management strategies. ‘Poached’ biomass is reported because it provides a direct indication of infringements under changing closure regimes, given a constant formulation of the infringement process. Discarded biomass is reported because a major management instrument in the RLF is a minimum legal size limit (MLS) for harvest (38cm TL, 36cm FL). Given that fish are selected by the gear below the MLS (Table 5), the relative level of discards provides feedback about potential unintended impacts of fishing because of changes in discard frequency under different management strategies. The objectives considered in this report and the relevant performance indicators are listed in Table 6.

Summary statistics for the numbers and perimeters of all reefs (gazetted and virtual) and areas of gazetted reefs closed and open to fishing under each closure regime are shown in Table 7. The numbers and perimeters of virtual reefs in each scenario are given in Table 8. The intended increases in closure regimes were not realised exactly, reaching only 26% and 40.1% of total reef perimeter closed for notional closures of 30% and 50% respectively. This under-representation of reef perimeters in closure regimes arose because the increases in closures were judged by the perimeter of the entire area closed, not the perimeters of the reefs within those areas. We do not consider this discrepancy between intended and actual increases in perimeters of closures particularly important since the figures of 30% and 50% were arbitrary and the realised actual increases in perimeters closed to fishing do represent significant increases over current closures (by ~1.6 and ~2.5 fold, Table 7).

Table 7: Aggregate numbers (N°) and **Perimeters** of all reefs (Gazetted + Virtual) and **Areas** of gazetted reefs **Open** and **Closed** to fishing under different **Closure Regimes**, respective percentages (%) of each variable in each category and percentage change from the current closures (%Cc) to alternative closure regimes. Note that virtual reefs were not assigned an area.

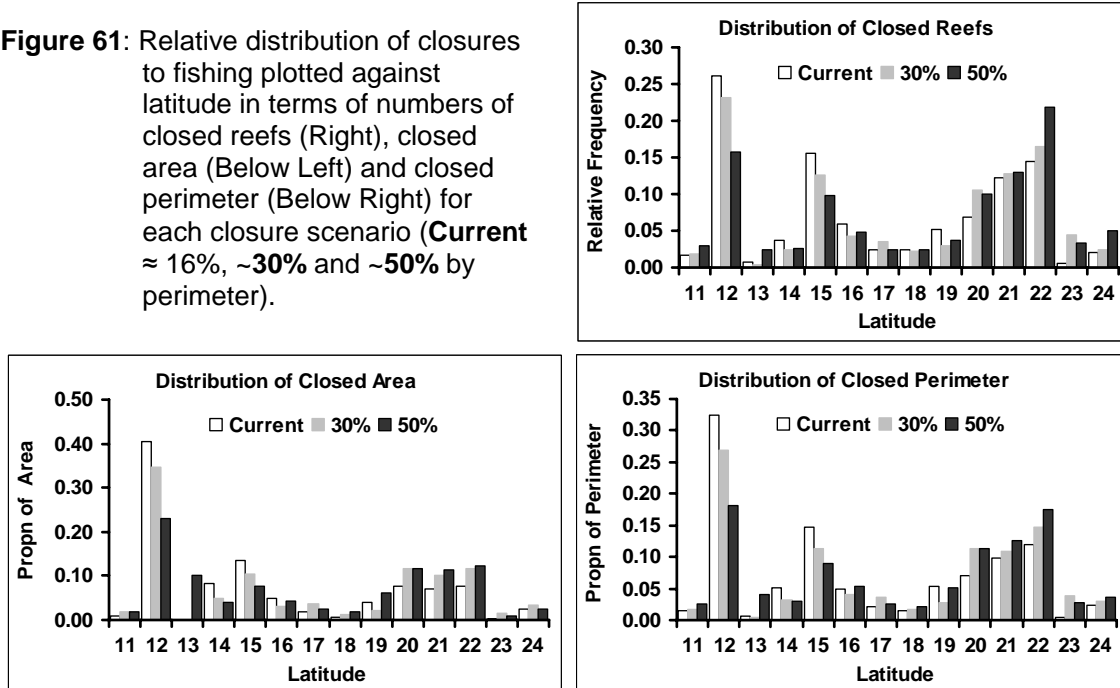
Closure		N° Reefs			Reef Area			Reef Perimeter		
Status	Regime	N°	%	% Cc	Km ²	%	% Cc	Km	%	% Cc
Open	0%	3,822	100		20,372	100		36,033	100	
	Current	3,240	84.8		15,877	77.9		30,211	83.8	
	30%	2,919	76.4	-9.9	12,609	61.9	-20.6	26,649	74	-11.8
	50%	2,398	62.7	-26.0	8,504	41.7	-46.4	21,566	59.9	-28.6
Closed	0%		0		0	0				
	Current	582	15.2		4,495	22.1		5,822	16.2	
	30%	903	23.6	155.3	7,762	38.1	172.7	9,384	26	161.2
	50%	1,424	37.3	244.7	11,868	58.3	264.0	14,467	40.1	248.5

Table 8: Numbers (N°) and **Perimeters** of virtual reefs **Open** and **Closed** to fishing under different **Closure Regimes**, respective percentages (%) of each variable in each category, percentage of all reefs that virtual reefs represented (**% All**), and percentage change in virtual reef characteristics from the current closures (**%Cc**) to alternative closure regimes. Virtual reefs were not allocated areas.

Closure		N° Reefs				Reef Perimeter			
Status	Regime	N°	%	% All	% Cc	km	%	% All	% Cc
Open	0%	1,182	100	31.0		10,522	100	29.2	
	Current	1,180	99.8	36.4		10,485	99.6	34.7	
	30%	1,158	98.0	39.7	-1.9	10,305	97.9	38.7	-1.7
	50%	1,120	94.8	46.7	-5.1	9,974	94.8	46.2	-4.9
Closed	0%	0	0			0	0		
	Current	2	0.2	0.3		37	0.4	0.6	
	30%	24	2.0	2.6	1,200	217	2.1	2.3	586.5
	50%	62	5.2	4.4	3,100	548	5.2	3.8	1,481

The distributions by latitude of reefs, reef area and reef perimeter closed under each closure regime are shown in Figure 61. Placing the additional closures under 30% and 50% regimes away from the large closure in the far north did result in greater balance between the far north and southern regions, but did not alter significantly the particularly low frequency of closures in the central latitudes (16°S and 19°S). This continued imbalance arose as a side effect of building additional closures around current closures, which are relatively few in these central latitudes (Fig. 61).

Figure 61: Relative distribution of closures to fishing plotted against latitude in terms of numbers of closed reefs (Right), closed area (Below Left) and closed perimeter (Below Right) for each closure scenario (**Current** ≈ 16%, ~30% and ~50% by perimeter).



We present all performance indicators relative to one of two reference points: an historic status, such as prior to the (presumed) start of the fishery (e.g., virgin biomass) or at a nominated time for which real data exist (e.g., target CPUE); or the status of the indicator under the default, or status quo, projection with current closures and 1996 levels of effort.

We present results of simulations in terms of these relative performance indicators in several ways. First, we present plots of the trajectories of selected performance indicators. Where it is informative, we present the trajectories for both initialisation and projection periods (*e.g.*, relative spawning biomass, relative available biomass). Where historical trajectories are either not of interest, irrelevant, or meaningless, we present trajectories only for the periods for which real data exist or the projection period (*e.g.*, CPUE, discarded biomass, poached biomass), or both. These trajectories display the dynamics of the performance indicator, especially during the transition from conditions constrained by the historical data to stable periods of future projections.

Second, we summarise the relative status of the performance indicators over the last five years of the projection period, and plot the value for each management scenario. These plots provide a perspective on the likely 'stable state' performance of the strategy set under consideration.

Third, where quantitative objectives were articulated by stakeholders and target values were stipulated for relevant performance indicators, we show the frequency with which those targets were likely to be met during the final five years of projections (summed over replicate projections for each scenario). These statistics indicate the likely degree to which different strategy sets will meet management objectives. They cannot be interpreted as the probability of realising the target in any year because of the correlations in the performance indicators among successive years within a replicate projection, but they are perhaps 'realistic' estimates of the likely performance of a management scenario, given that such temporal correlations in the behaviour of harvest and harvested populations are likely to be features of reality.

MSE Results

Results Summary

Most performance indicators were affected more by changes in effort levels than changes in amounts of area closed to fishing. Harvest-related objectives (e.g., maintaining CPUE, increased chance of catching a large fish, preserving available biomass) were most likely to be achieved when effort was lowest under any area closure strategy, but were less likely to be achieved as increasing amounts of area were closed to fishing. The principal stock-conservation objective, represented by preserving spawning biomass, was most likely to be achieved by increasing the amount of area closed to fishing and was only relatively slightly impacted by increasing fishing effort (relative to the 'status quo').

Most performance indicators varied substantially and conspicuously among projections under different sets of management strategies. Variation in the values of performance indicators among replicate runs of the same scenario stabilised after relatively few runs (8-10) and was consistently very small relative to the average value of performance indicators (CV ~1-3%). This result meant that there was little advantage to running more than 10 replicate projections of each scenario (Appendix C). The low variation among sets arose because the performance indicators were derived from data summed over hundreds or thousands of reefs. Whilst variation in most variables among reefs and through time (for individual reefs) was relatively large, it was not correlated over the entire spatial domain. Thus, the summed results tended to be relatively invariant⁸.

Simulation Results

Spawning Biomass

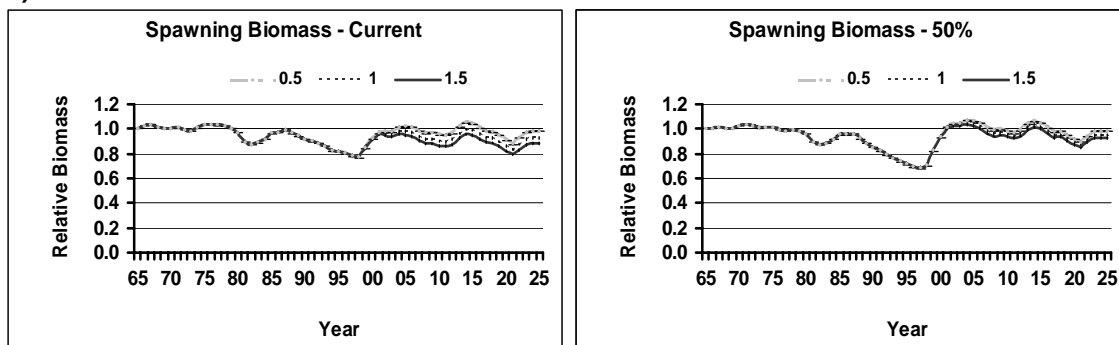
Spawning biomass responded rapidly to reef closures under all effort and closure scenarios, with relative spawning biomass rapidly reaching levels close to virgin levels and remaining so for most of the projection period (Fig. 62). Relative spawning biomass was slightly greater and impacts of fishing outside of closures slightly reduced at higher levels of closure (Fig. 62-A), but the effects of changing closure regime generally were slight.

Reducing effort to half of that in 1996 reversed the downward trend in spawning biomass for open reefs under all closure regimes and resulted in improved, stabilised levels relatively quickly (Fig. 62-B). Maintaining 1996 effort or allowing effort to grow resulted in sustained declines in spawning biomass on open reefs (Fig. 62-B). Changing closure regimes had only minor effects on the variation in trajectories of spawning biomass on open reefs caused by changing effort regimes (Fig. 62-B). Note, however, that the projections for each closure regime in Figure 62-B are for different sets of reefs and comparisons among the panels in the figure indicate only the extent to which changing closure regimes are likely to precipitate changes in the status of reefs left open to fishing compared to their prior status, rather than changes in the total spawning biomass *per se*.

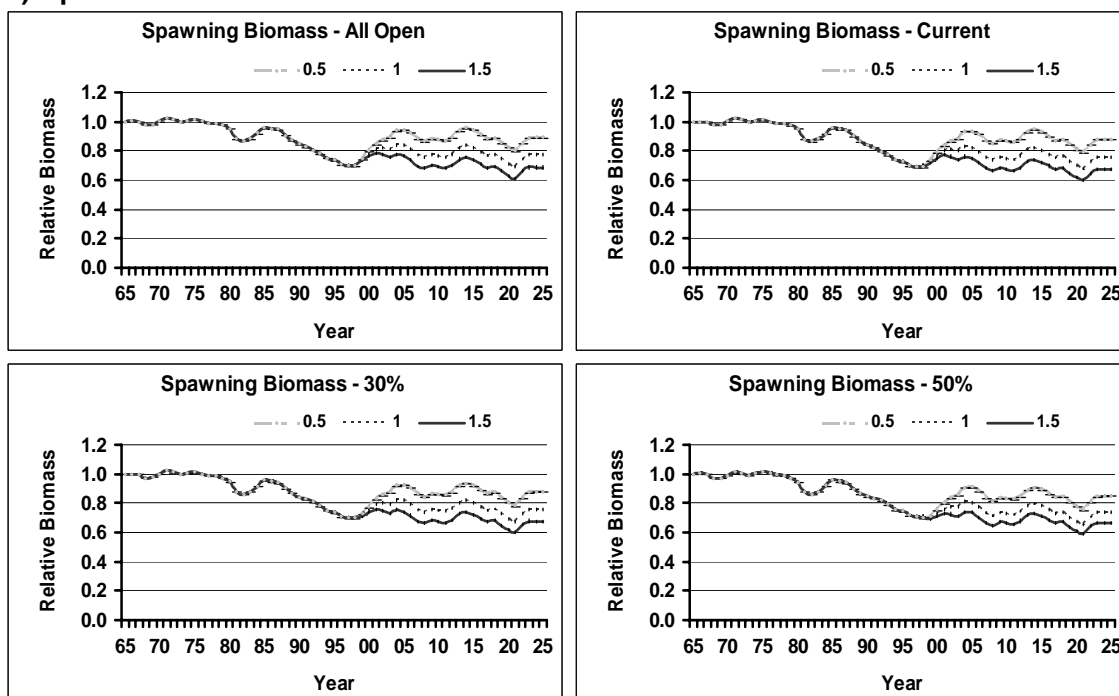
⁸ Because of the very low variation among replicate runs for the same management scenarios, error bars are not plotted on trajectories of variables presented below.

Figure 62: Trajectories spawning biomass relative to virgin spawning biomass averaged over ten simulations for reefs Closed (**A**) and Open (**B**) to fishing under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5** x1996 level). The interval 1965 – 1989 was the period of hind-casting, 1990-98 were the years of real data to which the simulations were tuned and the projection period was 1999-2025. Each strategy set was applied from 1999 to 2025.

A) Closed Reefs



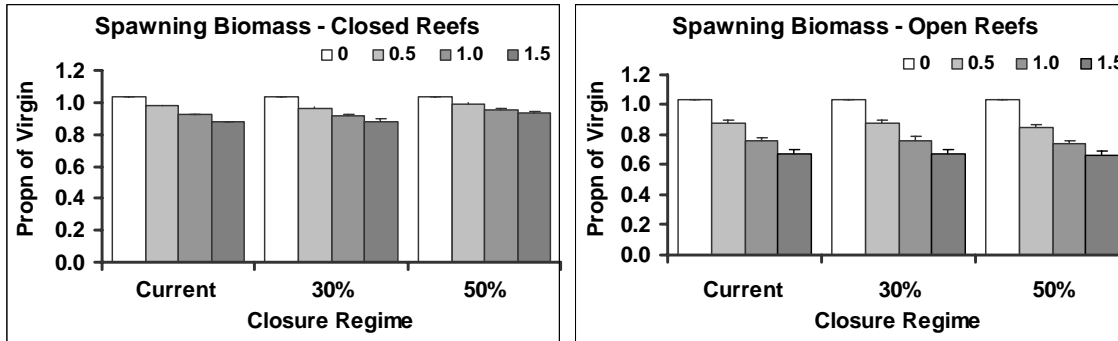
B) Open Reefs



Relative spawning biomass at the end of the projection periods on both closed and open reefs depended on the amount of fishing effort in open areas, though the effect was slight on closed reefs (<5%) (Fig. 63). Increasing the amount of reef closed to fishing improved the status of spawning biomass given each level of effort, thus attenuating the impacts of fishing in the system on the status of protected spawning biomass. This attenuation was at a maximum (~7%) when closures were increased from 30% to 50% under effort of 1.5 times that in 1996 (Fig. 63). Each step of 50% of 1996 effort reduced spawning biomass on open reefs by approximately 12-15% within each closure regime (Fig. 63). Changing the closure regime did not affect the impacts of changing effort on spawning biomass, substantially

attenuation of the impacts of a given level of effort by increasing closure being 1% or less (Fig. 63).

Figure 63: Mean spawning biomass over last five years of projection period (2021-25) as a proportion of virgin (unfished) spawning biomass on all reefs Closed (Left) and Open (Right) to fishing under four regimes of fishing effort (0, 0.5, 1.0 and 1.5 x 1996 level) and three regimes of area closure (% of reef perimeters; current ≈ 16%). Error bars are Standard Errors.

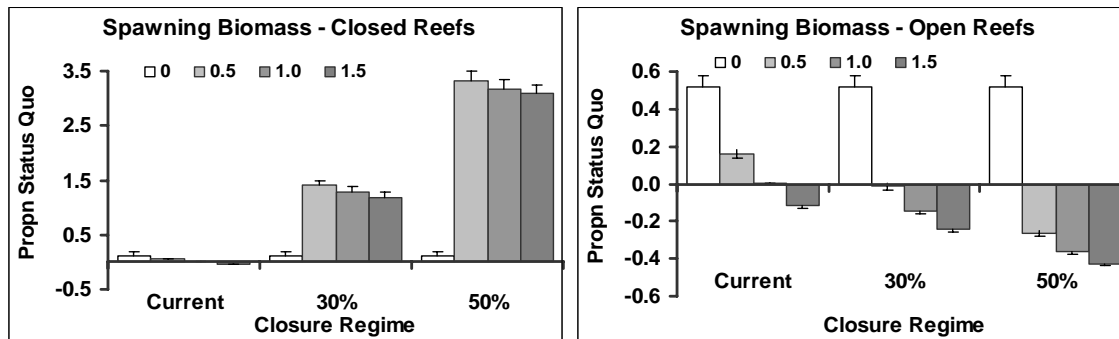


It is important to note that the above results arose from comparing the spawning biomass over 2021-2025 on a set of closed or open reefs with the spawning biomass on the same reefs prior to fishing. That is, comparisons were not made between different sets of reefs. When projected spawning biomass was considered relative to that projected under a default or 'status quo' scenario of current closures (16% of reef perimeters) and 1996 effective level of effort, however, greater differences among the management options were evident.

Increasing the amount of reefs closed to fishing dramatically increased the absolute amount of spawning biomass on closed reefs, irrespective of fishing effort (Fig. 64). This was expected simply because there were greater areas from which fishing was excluded. Note, however, that increases in spawning biomass on closed reefs exceeded the proportional increase in perimeter of reef closed (~1.6- and ~2.5-fold from current to 30% and 50% closures respectively) and the proportional increase in area closed (1.7- and 2.6-fold). For example, increasing closure from 16% to 30% of reef perimeter resulted in 2.2-2.4 fold increases in spawning biomass within the closed areas, and increasing closures to 50% (2.5-fold actual increase) resulted in 4.1-4.3 fold increases in spawning biomass protected within closed areas (Fig. 64).

Impacts of increasing closures and fishing effort on the spawning biomass in open areas relative to that expected under the status quo scenario also were consistent with expectation. Reducing effort under current closures, or eliminating effort altogether, resulted in substantially increased spawning biomass (by 15% and 50% respectively) in the open areas, whilst increasing effort resulted in an approximately 10% reduction in spawning biomass on open reefs (Fig. 64) compared to 'status quo' projections. Increasing closures effectively constrained effort to diminishing fishable areas, resulting in increased impacts on spawning biomass in those areas. These impacts also increased in magnitude with increasing effort (Fig. 64). Increasing closures to ~30% and ~50% by reef perimeter corresponded to actual reductions in fishable reefs by 11.8% and 28.6% by perimeter and 20.6% and 46.4% by area respectively (Table 7). These restrictions precipitated average reductions in spawning biomass of respectively 14.9% and 36.5% under a regime of 1996 effective effort and 24.4% and 42.7% when effective effort was increased to 1.5 times 1996 levels (Fig. 64). That is, only when effort increased was spawning biomass on reefs open to fishing reduced by what would be expected from the reductions in fishable area alone.

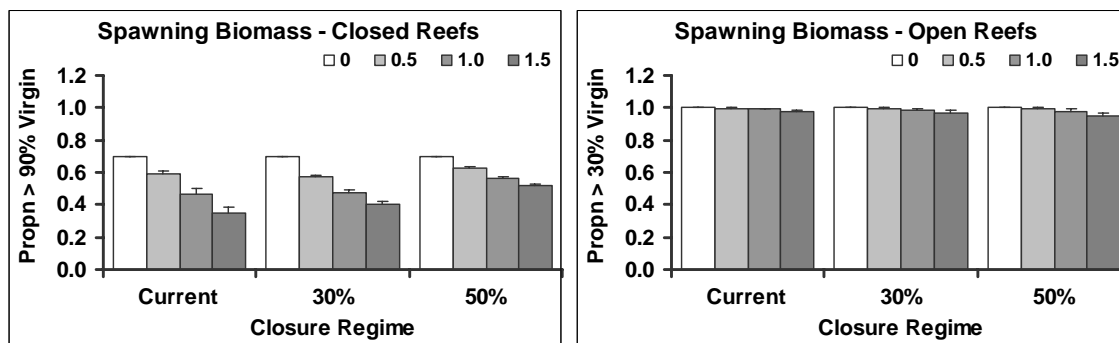
Figure 64: Mean spawning biomass over the last five years of the projection period (2021-25) relative to that realised under a 'status quo' projection (Current closures, 1996 effort) for reefs Closed (Left) and open (Right) to fishing under four regimes of fishing effort (**0**, **0.5**, **1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current \approx 16%). Values are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.



The proportion of reefs closed to fishing that realised spawning biomasses above 90% of virgin levels declined with increasing fishing effort (Fig. 65). The impacts of increasing effort were most marked under the current closure regime and were ameliorated by increasing the amount of reef closure (Fig. 65). Of particular interest was that even in the absence of fishing, on average only 70% of reefs would be expected to be above 90% of virgin spawning biomass in a given year (Fig. 65) and at 1996 effort levels, only about 50-56% of reefs would remain above 90% virgin spawning biomass. The apparently low percentage of closed reefs with greater than 90% of virgin spawning biomass is due to random variation in recruitment and natural mortality and the time for depleted reefs to recover even in the absence of exploitation.

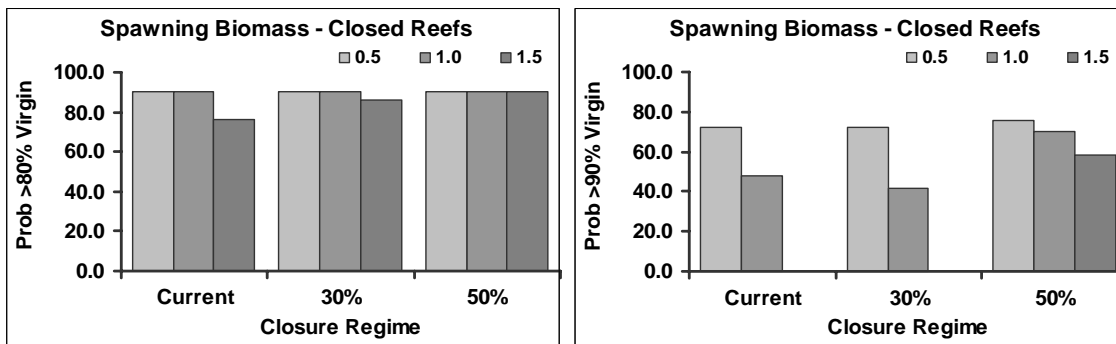
Increasing closure and effort each diminished only slightly the proportion of reefs open to fishing above 30% of virgin levels (Fig. 65). In general, spawning biomass on open reefs would fall below 30% of virgin spawning biomass on relatively few reefs (< 6%) in any of the scenarios simulated (Fig. 65).

Figure 65: Mean (over 2021-25) proportion of reefs closed to fishing with spawning biomass above 90% of virgin (unfished) spawning biomass (Left) and reefs open to fishing with spawning biomass above 30% of virgin (unfished) spawning biomass (Right) during the last five years of projections under four regimes of fishing effort (**0**, **0.5**, **1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current \approx 16%). Error bars are Standard Errors.



The performance indicators and targets for closed reefs were: “spawning biomass in areas closed to fishing should remain above 80% of virgin spawning biomass for 90% or more of the time, and preferably close to 90% of virgin spawning biomass”. There were significant effects of both changing effort and changing closure regimes on the achievement of these performance targets for spawning biomass on closed reefs (Fig. 66). Projected overall biomass on closed reefs exceeded 80% of virgin spawning biomass on those reefs 90% of the time in all closure scenarios when effort was half or equal to that in 1996 (Fig. 66). When effort was set at 1.5 times that in 1996, however, the minimum performance target was met only when 50% of habitat (reef perimeter) was closed to fishing (Fig. 66). The frequency with which the desired target of 90% of virgin spawning biomass on closed reefs would be realised decreased substantially with increasing fishing effort within each closure regime and generally increased with increasing amount of reef closed to fishing, particularly when effort levels were highest (Fig. 66). Increasing the amount of reef closed compensated for the effects of increasing effort most at higher effort levels (Fig. 66).

Figure 66: Percentage of the last five years of the projection period (2021-25) for 10 simulations when spawning biomass was above 80% (Left) and 90% (Right) of virgin (unfished) spawning biomass on reefs Closed to fishing under three regimes of fishing effort (0.5, 1.0 and 1.5 x 1996 level) and three regimes of area closure (% of reef perimeters; current ≈ 16%).

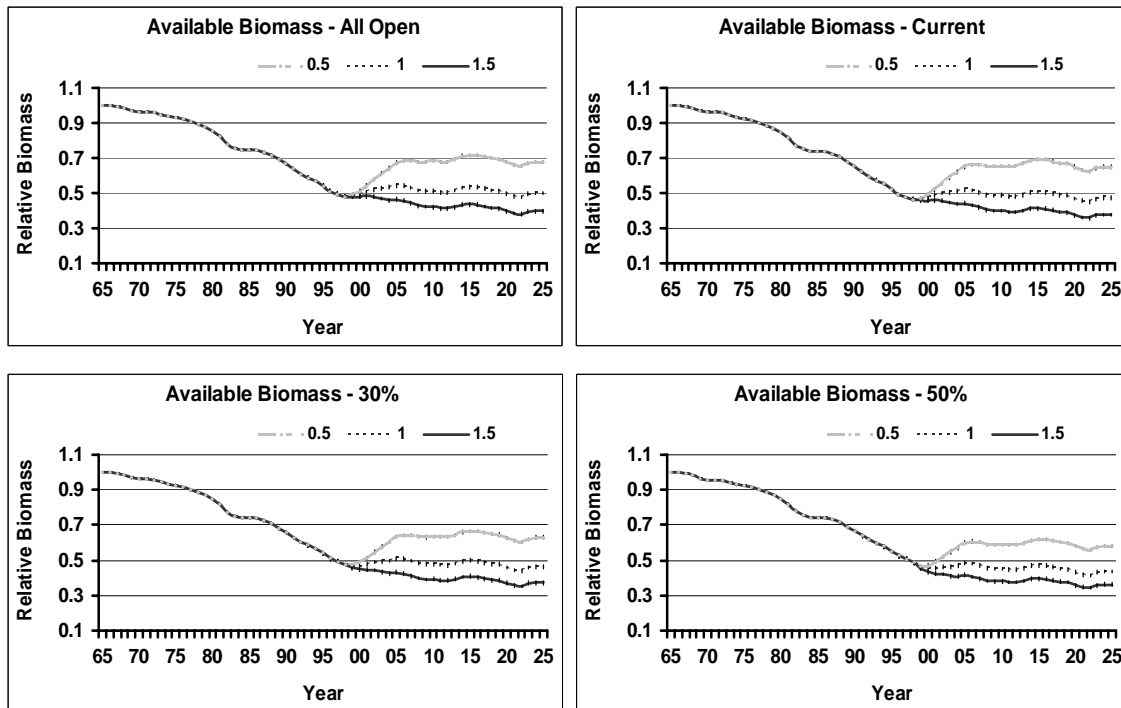


Available Biomass

Trajectories of available biomass (fish above the minimum legal size and selected by the gear) were similar in character to those for spawning biomass under the different management scenarios. For all closure regimes, reducing effective effort to half 1996 levels resulted in fairly rapid increase in relative available biomass that stabilised after 5-6 years of projection (Fig. 67). Maintaining effort at the 1996 levels resulted in little change in available biomass from that at the end of the ‘historical’ period (~45% in 1998), although there were very slight continuing downward trends under the 30% and 50% closure regimes (Fig. 67). Allowing effort to increase resulted in continued depletion of available biomass, although the trajectories stabilised at around 35% of virgin levels in all closure regimes (Fig. 67). Increasing closures depressed the trajectories of available biomass slightly under reduced and ‘status quo’ effort, but had little effect under the increased effort scenario (Fig. 67).

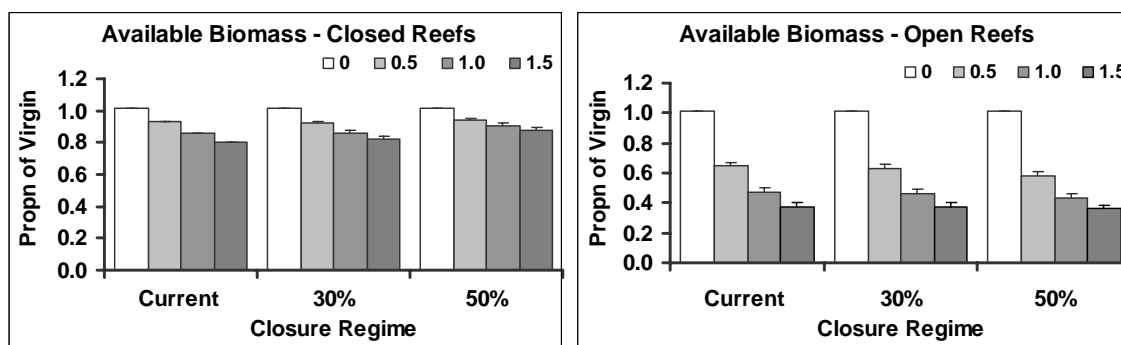
Available biomass exceeded 50% of virgin available biomass in all closure regimes under the reduced effort scenario, but otherwise reached 50% only when effort was held at 1996 levels and closures were at or below current levels (Fig. 67). Available biomass on open reefs approached or exceeded 70% of virgin available biomass when fishing was present only if effort was reduced to 50% of 1996 levels and closures were maintained at current levels or less (Fig. 67).

Figure 67: Trajectories of mean available biomass (legal sized fish, > 38cm TL) relative to virgin available biomass for reefs Open to fishing under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three effort regimes (**0.5**, **1.0** and **1.5** x 1996 level). The interval 1965–89 was the period of hind-casting, 1990-98 were the years of real data to which the simulations were tuned and the projection period was 1999-2025. Each strategy set was applied from 1999.



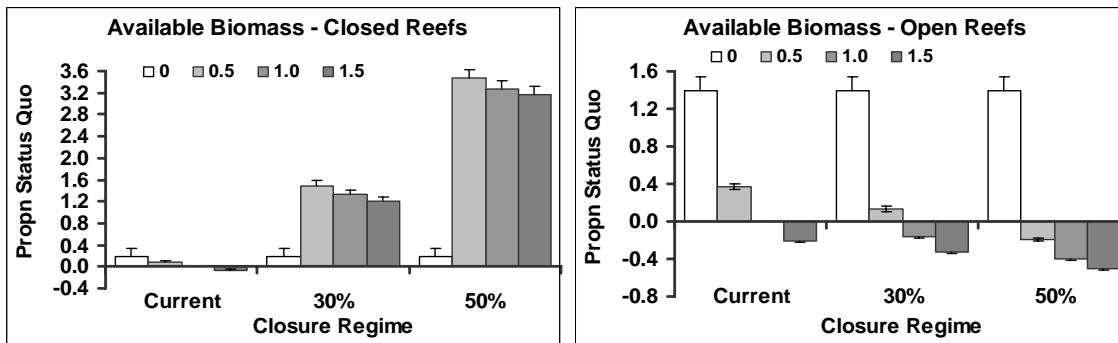
Available biomass at the end of the projection period on closed reefs remained high relative to un-fished levels in all closure and effort scenarios. Only slight improvements were gained through increasing amounts of closure (Fig. 68). Increasing effort significantly reduced available biomass on the open reefs under each of the closure strategies, and again increasing closure had only marginal effects on the impacts of changing effort (Fig. 68).

Figure 68: Mean available biomass over the last five years of the projection period (2021-25) as a proportion of virgin (unfished) available biomass on all reefs Closed (Left) and Open (Right) to fishing under four regimes of fishing effort (**0**, **0.5**, **1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current ≈ 16%). Error bars are Standard Errors.



As expected, increasing closures dramatically increased the available biomass in closed areas over the status quo projections, primarily because of the greater amount of habitat subject to low or zero fishing pressure (Fig. 69). As with spawning biomass, increasing effort slightly diminished the effectiveness of increasing closures (Fig. 69). Increasing closures had the reverse effect on reefs open to fishing, resulting in diminished available biomass relative to status quo projections for given levels of effort (Fig. 69). Increasing effort exacerbated this effect, further diminishing available biomass within each closure regime (Fig. 69). Available biomass compared with status quo projections for open reefs was reduced by 17% and 40% for 1996 levels of effort as closures rose to 30% and 50% respectively. These reductions were roughly the same as the proportional reduction in areas of fishable reef (~20% and ~46% respectively) but considerably more than the reductions in reef perimeter open to fishing (approximately 12% and 29% respectively). Increasing effort to 1.5 times 1996 levels under the current closure regime resulted in available biomass in open areas falling by ~20%. Available biomass was reduced further to 33% and 50% as closure regimes were incremented and so the amount of fishable ground fell (Fig. 69). These impacts were considerably greater than projected from changes in perimeter alone, but still less than might be expected from the combined effects of reduced area and increased effort.

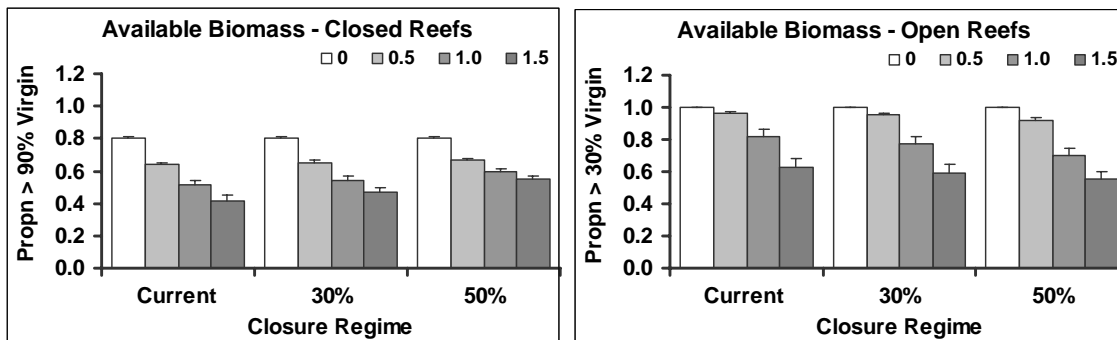
Figure 69: Mean available biomass over the last five years of the projection period (2021-25) relative to that expected under a 'status quo' projection (Current closures, 1996 effort) for reefs Closed (Left) and open (Right) to fishing under four regimes of fishing effort (0, 0.5, 1.0 and 1.5 x 1996 level) and three regimes of area closure (% of reef perimeters; current ≈ 16%). Values are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.



The biomass of legal sized coral trout was above 90% of virgin levels on around 80% of closed reefs in the absence of fishing. This proportion fell progressively with increasing fishing effort in the system, reaching a minimum of slightly over 40% of reefs when effective effort was 1.5 times that in 1996 under current closure regimes (Fig. 70). The impacts of increased fishing diminished with increasing amounts of closure (Fig. 70), indicating that the greater (and larger) closures buffered the direct or indirect impacts of fishing on the 'protected' populations of fish.

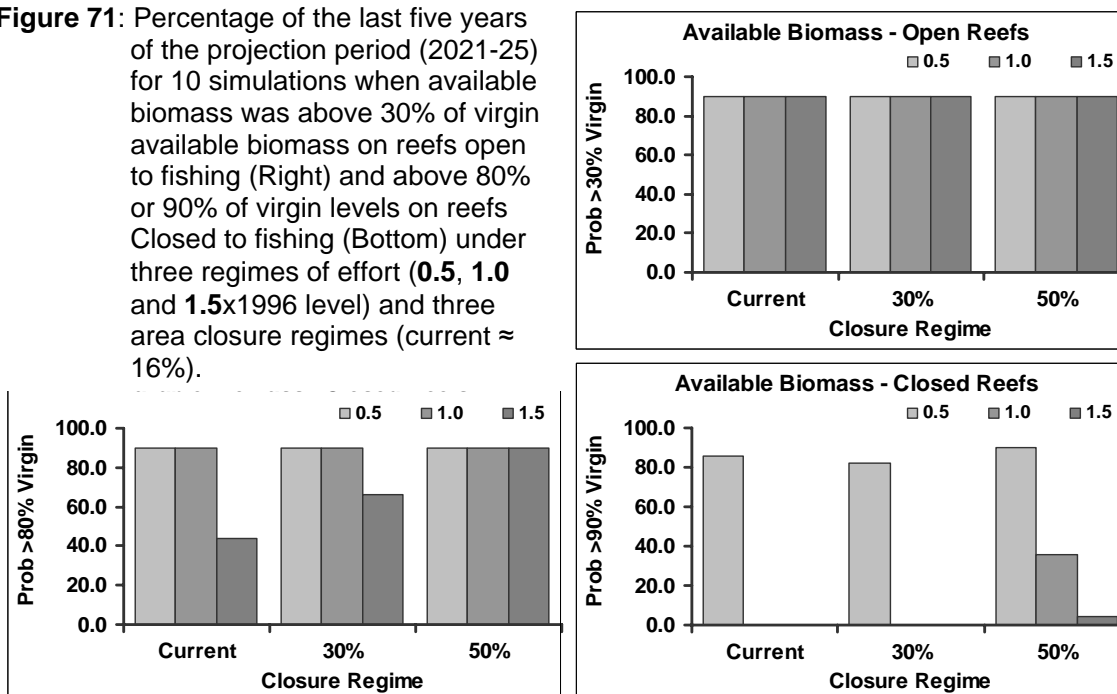
As with spawning biomass, the effect of increasing closures was reversed for the areas open to fishing. That is, the proportion of open reefs on which available biomass would remain above 30% of virgin available biomass decreased as a result of increasing closures for each effort scenario (Fig. 70). The effects of increasing effort, however, were far more severe than increasing closures, with available biomass remaining above 30% of the virgin level on around 85-90% of open reefs when effort was at 0.5 of the 1996 levels but only 55-62% of reefs when effort was allowed to grow to 1.5 times 1996 levels (Fig. 70).

Figure 70: Mean proportion of closed reefs with available biomass above 90% of virgin (unfished) available biomass (Left) and open reefs with over 30% virgin available biomass (Right) during the last five years (2021-25) of the projection period under four effort regimes (0, 0.5, 1.0 and 1.5 x 1996 level) and three closure regimes (% of reef perimeters; current ≈ 16%). Error bars are Standard Errors.



The performance objective for available biomass on open reefs (available biomass in areas open to fishing being above 30% of virgin available biomass 90% of the time) would be likely to be met under all of the management scenarios examined, with available overall biomass expected to remain above 30% of virgin level in 9 out of 10 years under each strategy set (Fig. 71). Aggregate available biomass on closed reefs would be expected to remain above 80% of virgin levels under all strategy sets except those involving increased effort and current or 30% closure regimes (Fig. 71). Available biomass on the closed reefs would be highly likely to remain above 90% of virgin, however, only with fishing effort at half of the 1996 level (Fig. 71).

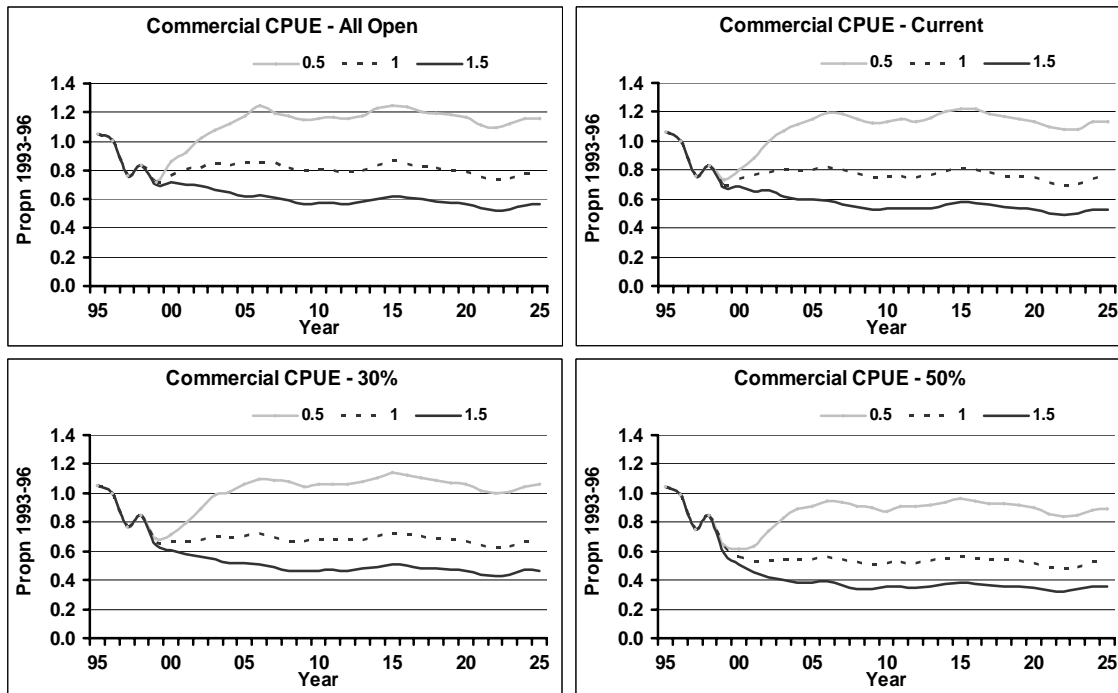
Figure 71: Percentage of the last five years of the projection period (2021-25) for 10 simulations when available biomass was above 30% of virgin available biomass on reefs open to fishing (Right) and above 80% or 90% of virgin levels on reefs Closed to fishing (Bottom) under three regimes of effort (0.5, 1.0 and 1.5x1996 level) and three area closure regimes (current ≈ 16%).



Commercial Fishery Performance

Trajectories of commercial catch per unit of fishing effort (CPUE) indicated that maintaining effective effort at 1996 levels would result in CPUE remaining relatively static over the projection period at around 80% of what it had been between 1993 and 1996 (Fig. 72). Note that CPUE had already dropped to slightly (~10%) below the 1993-96 average by the time the projections began in 1999. Increasing or decreasing effort resulted in a period of adjustment in CPUE lasting approximately 5-10 years followed by relatively constant trajectories at levels respectively approximately 40% below or 20% above the reference value (Fig. 72). These changes in CPUE represented respectively a slight reduction (by ~15%) and considerable enhancement (by ~70%) from the status of commercial CPUE at the beginning of the projection period (Fig. 72). Changing the amount of area closed to fishing had negligible effects on CPUE trajectories (Fig. 72).

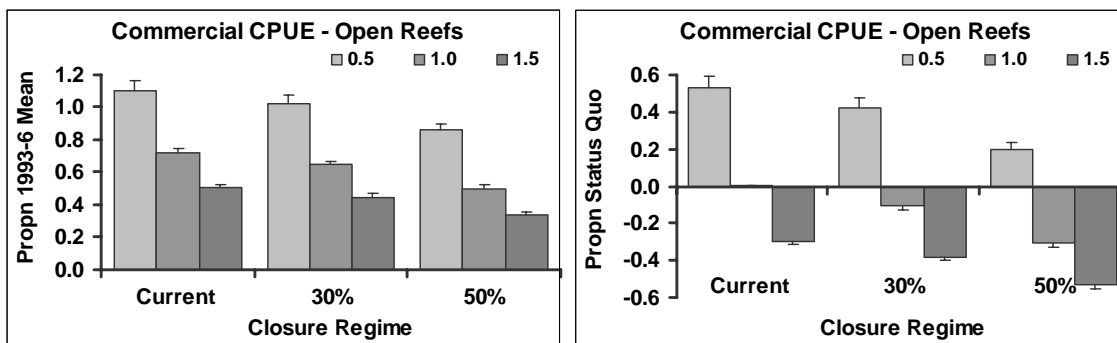
Figure 72: Trajectories of mean catch per unit of effort (CPUE) by commercial fishers relative to average CPUE between 1993 and 1996 over ten simulations for reefs open to fishing under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5** x 1996 level). The interval 1995 – 1998 are years of real data with trajectories thereafter being projections under each strategy set.



As expected from the above trajectories, commercial CPUE at the end of projections relative to average CPUE historically (1993-96) varied greatly with effort within each closure regime but was affected only slightly by changing the amount of closed reef under a given level of effort (Fig. 73). Halving effort under current closures resulted in CPUE exceeding that historically and being more than 50% better than projected from current effort and closure regimes (Fig. 73). Even with effort static at 1996 the level, projected CPUE would be predicted to be only around 75% of the 1993-96 average under current closures and slightly less if closures were increased (Fig. 73). Increasing effort by 50% led to greater diminution of relative CPUE, with projections being 25-30% less at the end of the projection period than for the 1996 levels of effort and around 40% of the 1993-96 average catch rates (Fig. 73).

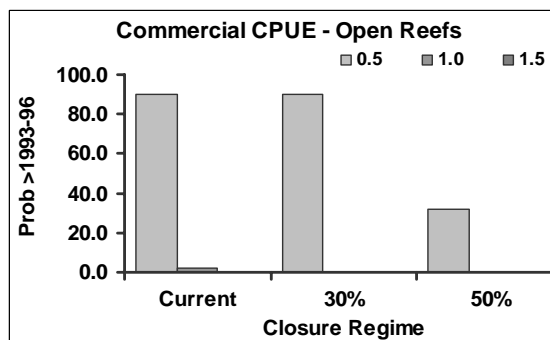
The interaction between closure regime and amount of effort became clearer when projected CPUE was compared to that expected under projected status quo conditions. Increases in CPUE that would accrue through reducing effort would be eroded as closures were increased and fishable area decreased. For example, the gains from halving effort fell from around 55% under current closures to less than 20% with ~50% of reef perimeters closed to fishing (Fig. 73). Conversely, the penalties to CPUE arising from increased effort were exacerbated by increasing closures. For example, CPUE fell by approximately 30% when effort was increased by 50% under current closures, but fell by over 50% when closures were increased to 50% of reef perimeters (Fig. 73).

Figure 73: Mean CPUE of commercial fishers during the last five years of the projection period (2021-25) relative to the average commercial CPUE in the period 1993-96 (Left) and relative to that realised under a 'status quo' projection (Current closures, 1996 effort) (Right) from reefs open to fishing under three regimes of fishing effort (**0.5**, **1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current \approx 16%). Values in the right figure are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.



The objective target for commercial CPUE was that CPUE remain above 80% of the mean CPUE derived from compulsory logbook records between 1993 and 1996. This target was likely to be met consistently only if effort was reduced from the 1996 level and would be unlikely to be met even then if areas closed to fishing were increased to 50% (Fig. 74). In contrast, increasing closure from current levels (~16%) to 30% of reef perimeters had negligible impact on the likelihood that the objective target would be realised (Fig. 64).

Figure 74: Percentage of the last five years of the projection period (2021-25) for 10 simulations when annual average commercial CPUE was above 80% of the 1993-96 average on reefs open to fishing under three regimes of effort (**0.5**, **1.0** and **1.5** x 1996 level) and three closure regimes (% of reef perimeters; current \approx 16%).



Projected commercial harvest from open reefs changed with changing levels of effort but the relationship was not linear. Under current or zero closure regimes, total harvest from open reefs relative to that in 1996 decreased substantially (by ~25%) with reduced effort but increased only slightly (~5%) with increased effort (Fig. 75). Relative harvest decreased as

closure regimes increased and the effects of changing effort were substantially diminished as the amount of closure was increased (Fig. 75). Notably, commercial harvest was almost always less than that in 1996, even under status quo projections, largely because the harvest of coral trout in 1996 was one of the highest recorded (QFS 2002).

Figure 75: Trajectories of commercial harvest from open reefs relative to reported harvest in 1996 from reefs open to fishing then averaged over ten simulations for reefs open to fishing under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5** x 1996 level). The interval 1990 – 1998 are years of real data with trajectories thereafter being projections under each strategy set.

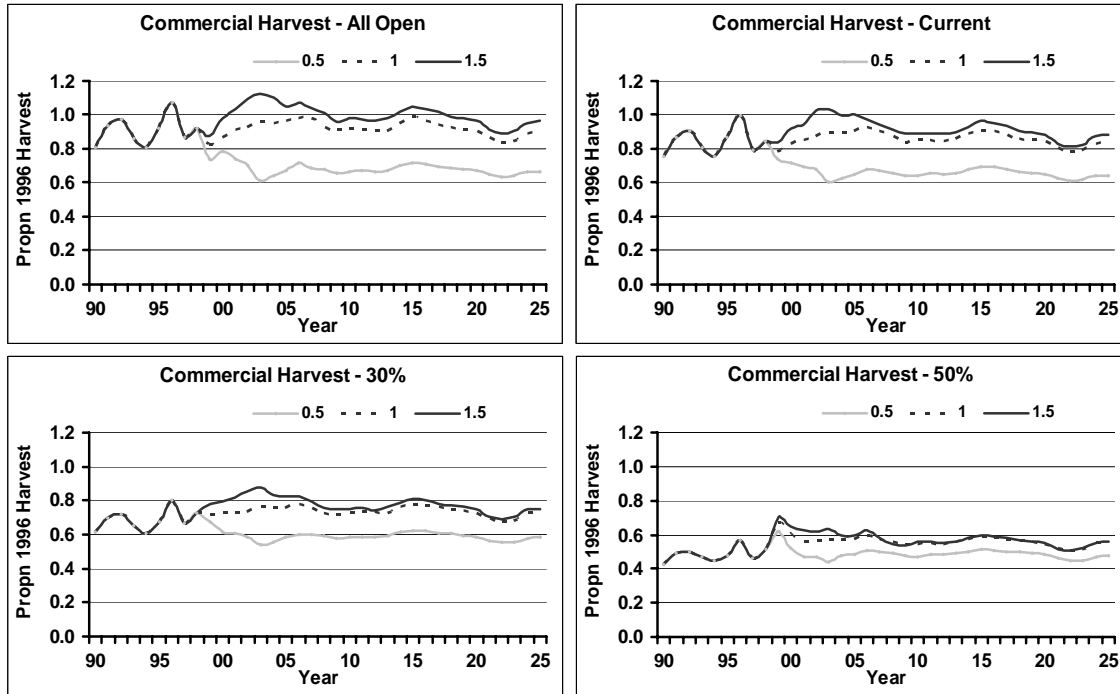
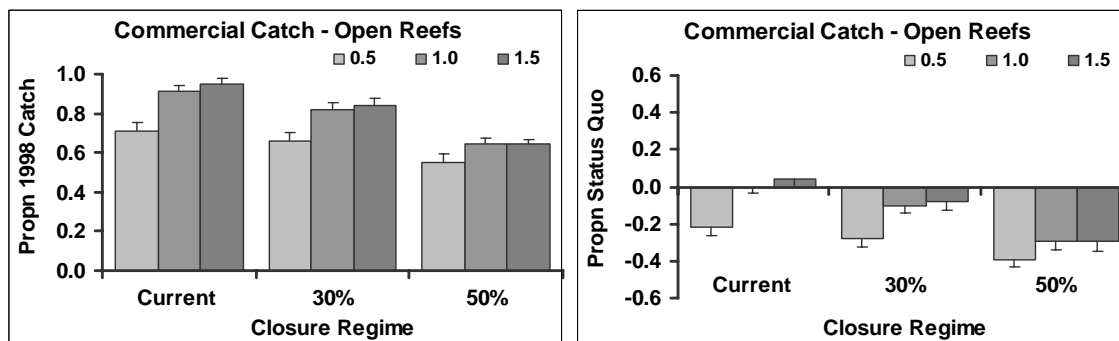


Figure 76 shows average annual harvest over the last five years of projections (2021-25) relative to a common index – total reported harvest in 1998, at the end of the initialisation period. Total harvest fell when effort was halved and fell with increasing closures (Fig. 76). Neither of these reductions, however, was in proportion to the reduction of either effort or fishing ground. Reductions in harvest attributable to a 50% effort reduction ranged from approximately 22% under current closures to only 14% in the 50% closure regime (Fig. 76). Total commercial catch of coral trout also increased with increasing effort, but only very slightly (5%) under current closures and not at all when closures were increased to 50% of reef perimeters (Fig. 76). With effort constant at 1996 levels, projected total commercial harvest in 2021-25 declined from a high of 91% (of 1998 catch) under current closures to a low of 64% of the 1998 catch when 50% of reef perimeters were closed to fishing.

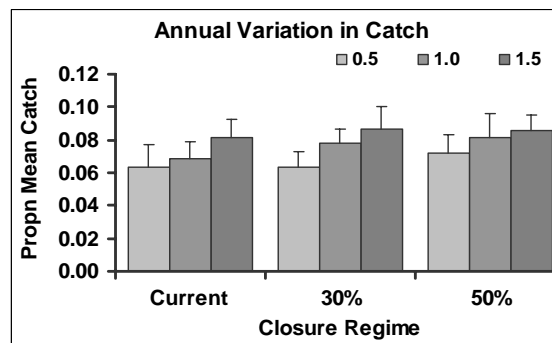
Projected catch for each management set was never substantially greater than that projected under status quo conditions and was usually was substantially less, because of either decreased effort or decreased fishing ground (and so available biomass), or both (Fig. 76). Reductions in total catch resulting from increasing closures exceeded any increase in catch resulting from increased effort and significantly reduced the total harvest taken under any level of effort (Fig. 76). Thus, total commercial harvest would be maximised by allowing increased fishing effort under the current closure regime, although even under these conditions the total harvest was not predicted to exceed that taken in 1998 (Fig. 76).

Figure 76: Mean annual commercial harvest during the last five years of the projection period (2021-25) relative to the reported catch in 1998 (Left) and relative to that realised under a 'status quo' projection (Current closures, 1996 effort) (Right) from reefs open to fishing under three regimes of fishing effort (**0.5**, **1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current \approx 16%). Values in the right figure are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.



Although the estimates of year-to-year fluctuations in total commercial harvest consistently increased with increasing effort and closure, none of these apparent changes would be deemed statistically significant (Fig. 77). Thus, uncertainty in catch would seem relatively unaffected by the management strategies we considered and ranged from 6% to 8% of annual harvest across all scenarios (Fig. 77).

Figure 77: Inter-annual fluctuations in annual commercial catch over the last five years of the projection period (2021-25) relative to the mean catch over the same period from reefs open to fishing under three regimes of fishing effort (**0.5**, **1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current \approx 16%). Error bars are Standard Errors.

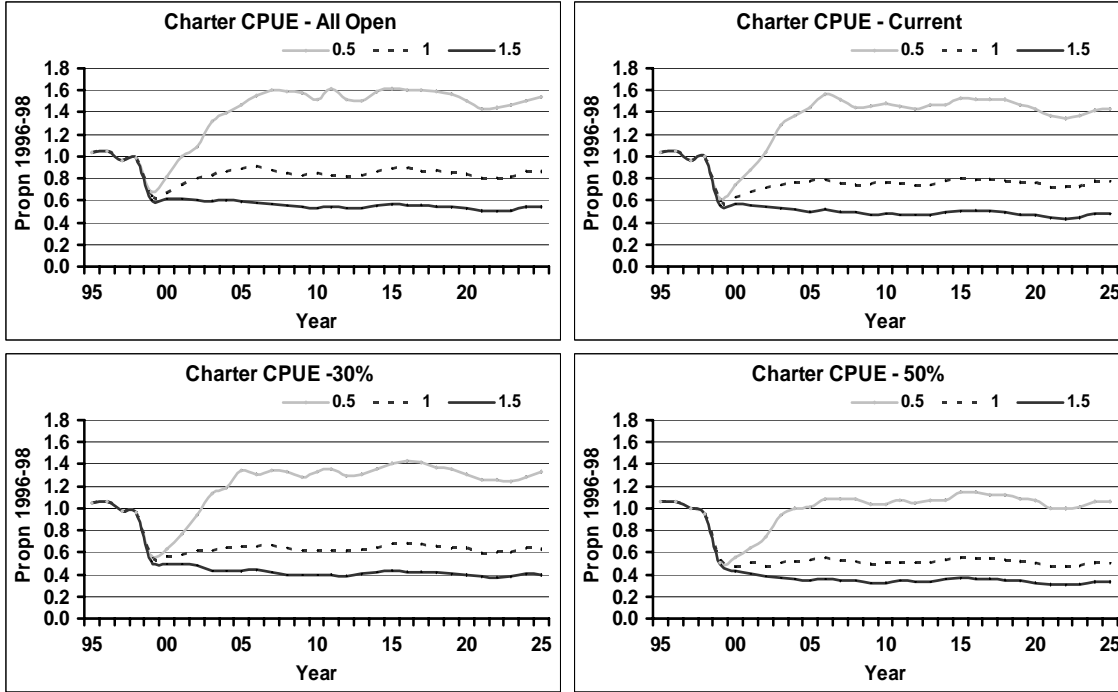


Charter and Recreational Fishery Performance

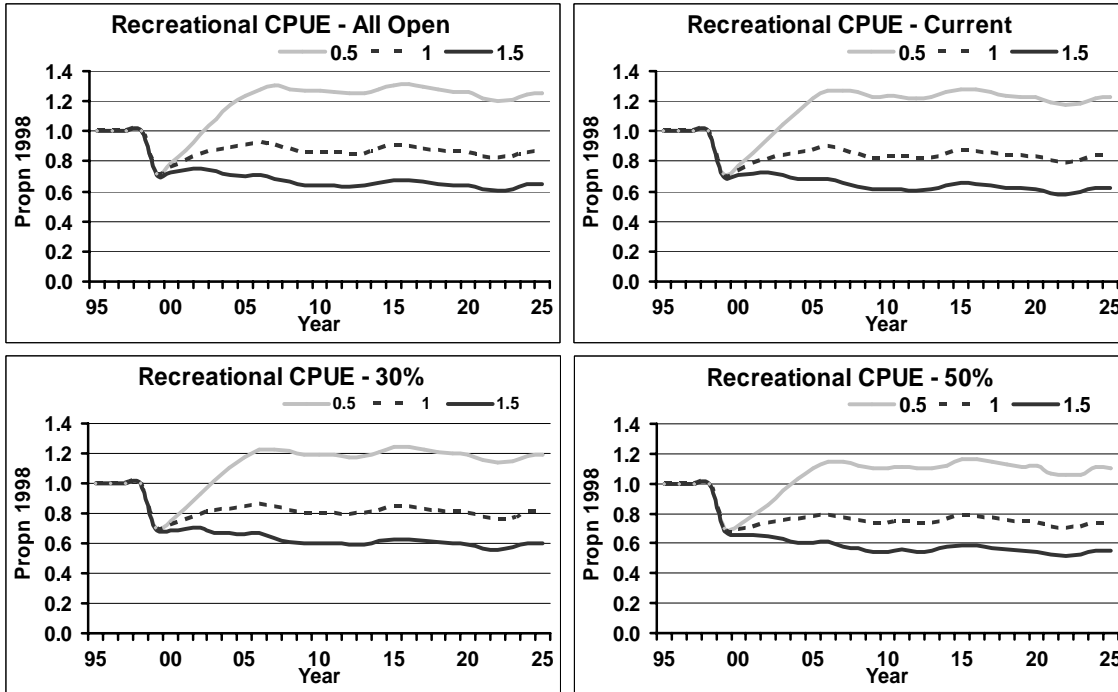
Responses of charter and recreational CPUE were qualitatively similar to those for the commercial sector, although both recreational and charter CPUE declined sharply in the first year of the projections (Fig. 78). Reducing effort provided significantly and substantially elevated projections of CPUE over either leaving effort at 1996 levels or allowing it to increase by 50% (Fig. 78, Fig. 79). CPUE responded to halving effort over the first 5-10 years of projections to stabilize thereafter at levels at or substantially above those derived from logbooks or surveys in the late 1990s (Fig. 78). The initial sharp drops in CPUE during projections were maintained when effort was set at 1.5 times that in 1996, whilst holding effort at 1996 levels resulted in only slight improvements over CPUE early in the projection period and did not result in projected CPUE ever recovering to those estimated from available data (Fig. 78).

Figure 78: Trajectories of catch per unit of effort (CPUE) by charter fishers (A) relative to average CPUE between 1996 and 1998 and recreational fishers (B) relative to reported CPUE in 1998 averaged over ten simulations for reefs open to fishing under four closure regimes (All open, 16% (Current), 30% and 50% closed) and three regimes of effort (0.5, 1.0 and 1.5 x 1996 level).

A) Charter Fishing Relative CPUE



B) Recreational Relative CPUE



It was clear that reducing effort provided substantially better outcomes at the end of the projections than either of the other effort control strategies (Fig. 79). CPUE exceeded the

historical reference levels only if effort was reduced. Under projections of 1996 or greater levels of effort, recreational and charter CPUE was less than 65% of historical levels (Fig. 79). As with commercial CPUE, the benefits of reducing effort were decreased and the impacts of increasing effort were exacerbated by increasing the amount of area closed to fishing (Fig. 79). The impacts of increasing effort (of all sectors) and increasing closures on historically indexed CPUE were more dramatic for charter fishers than for either recreational fishers (Fig. 79) or commercial fishers (Fig. 73.)

Similarly, differences between status quo and other projections were more extreme for charter fishers' CPUE than for either recreational or commercial CPUE, although the character of the results was consistent across all sectors (Fig. 79, Fig. 73). Notably, the impacts of changing closure regimes on endpoints of projections relative to those from the status quo scenario were relatively minor for recreational fishers (Fig. 79).

Figure 79: Mean CPUE of charter fishers (Top) and recreational fishers (Bottom) during the last five years of the projection period (2021-25) relative to the average CPUE in the period 1996-98 (Charter) or 1998 (Recreational) (Left) and relative to that realised under a 'status quo' projection (Current closures, 1996 effort) (Right) from reefs open to fishing under three regimes of fishing effort (**0.5**, **1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current \approx 16%). Values in the right figure are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.

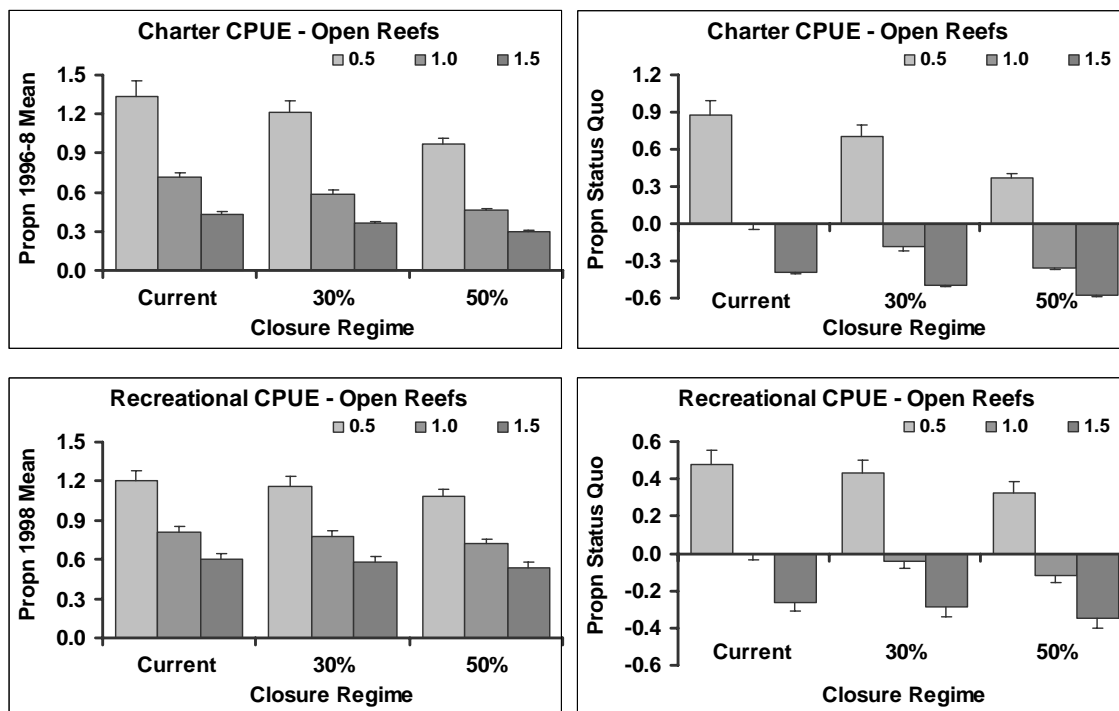
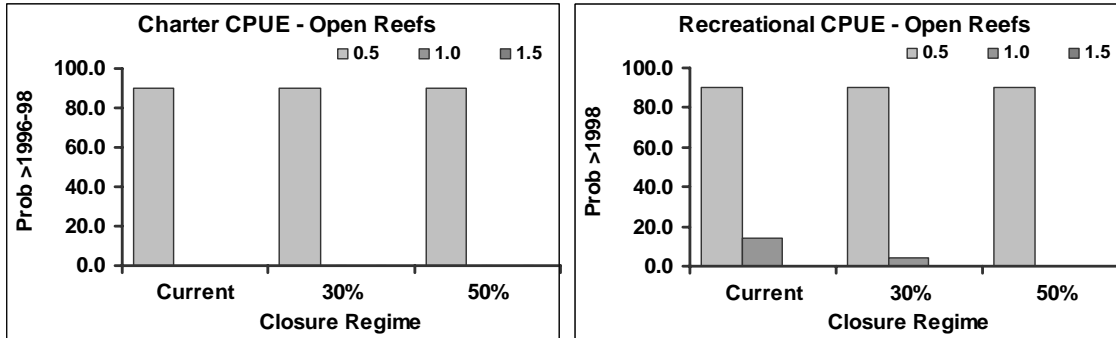


Figure 80 shows the results of applying a similar objective target to charter and recreational fishing catch rates as that applied to commercial fishing. As with commercial fishing, the objective was likely to be met only under conditions of reduced fishing effort, irrespective of closure regime (Fig. 80).

Figure 80: Percentage of the last five years of the projection period (2021-25) for 10 simulations when annual average charter CPUE (left) and recreational CPUE (right) was above 80% of the 1996-98 (charter) or 1998 (recreational) average on reefs open to fishing under three regimes of effort (**0.5, 1.0** and **1.5** x 1996 level) and three closure regimes (% of reef perimeters; current ≈ 16%).



Relative total catches by both the recreational and charter sectors varied with effort and closure regime, but in patterns qualitatively and quantitatively different from those for the commercial sector and from each other.

Recreational catches plummeted rapidly in the early years of the projections but recovered rapidly over the following 5-10 years under status quo and increased effort scenarios (Fig. 81). Reduced effort resulted in continued decline in catches, which stabilised well below those estimated in 1996 and those resulting from the initial fall in the early years of projections (Fig. 81). Changing closure regimes had relatively slight impacts on either the amount of catch taken relative to 1996 (~5% depreciation with each step in effort) or on the effects of changing fishing effort (Fig. 81).

Averaged over the last five years of the projection period, recreational catches decreased with decreased effort and increased marginally with increased effort, though by considerably less than the magnitude of effort increase (Fig. 82), similar to catches from commercial fishers (Fig. 76). Changing closure regimes, however, had negligible effect on either relative recreational catches or on the magnitudes of effects of different levels of effort (Fig. 82). This differed markedly from the effects of changing closure regime on catches by the commercial fishery (Fig. 76). Thus, projected catches relative to those under status quo conditions varied (or remained static) in accordance with variations in effort, decreasing effort resulting in diminished total harvest and increased effort resulting in increased harvest relative to status quo conditions (Fig. 82).

Figure 81: Trajectories of recreational harvest from open reefs relative to estimated harvest from open reefs in 1996 averaged over ten simulations under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5** x 1996 level). 1998 was the single year of real data and trajectories thereafter are projections under each strategy set.

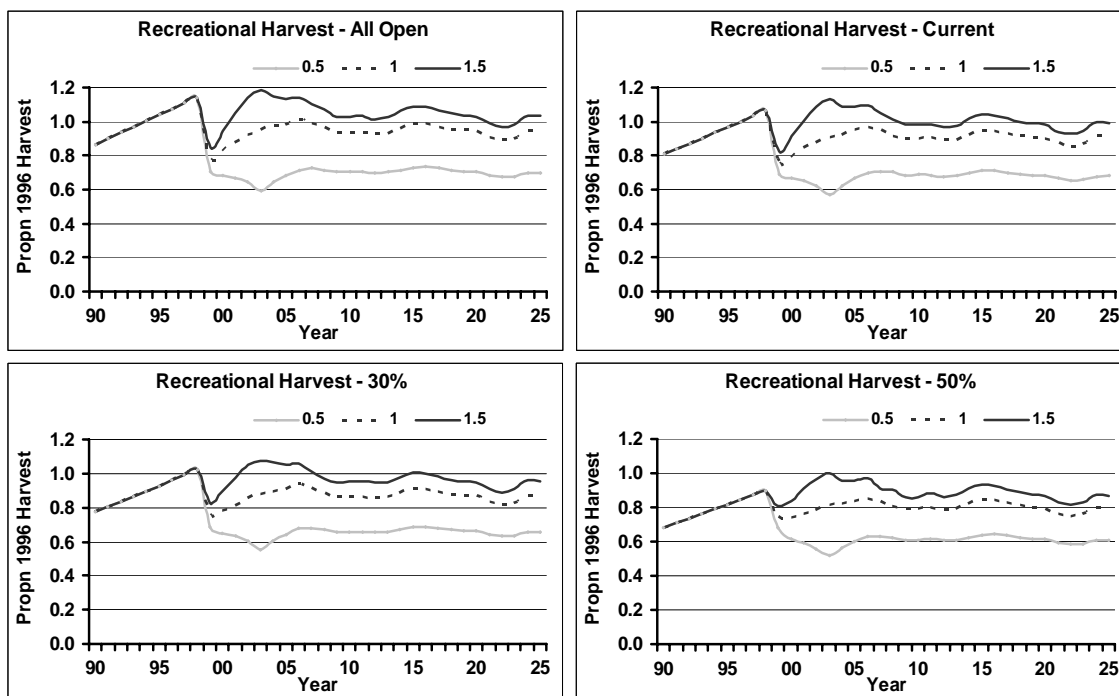
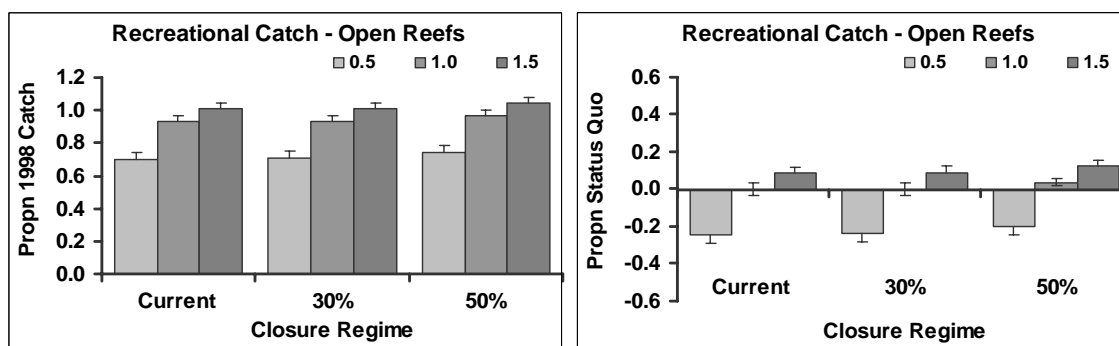
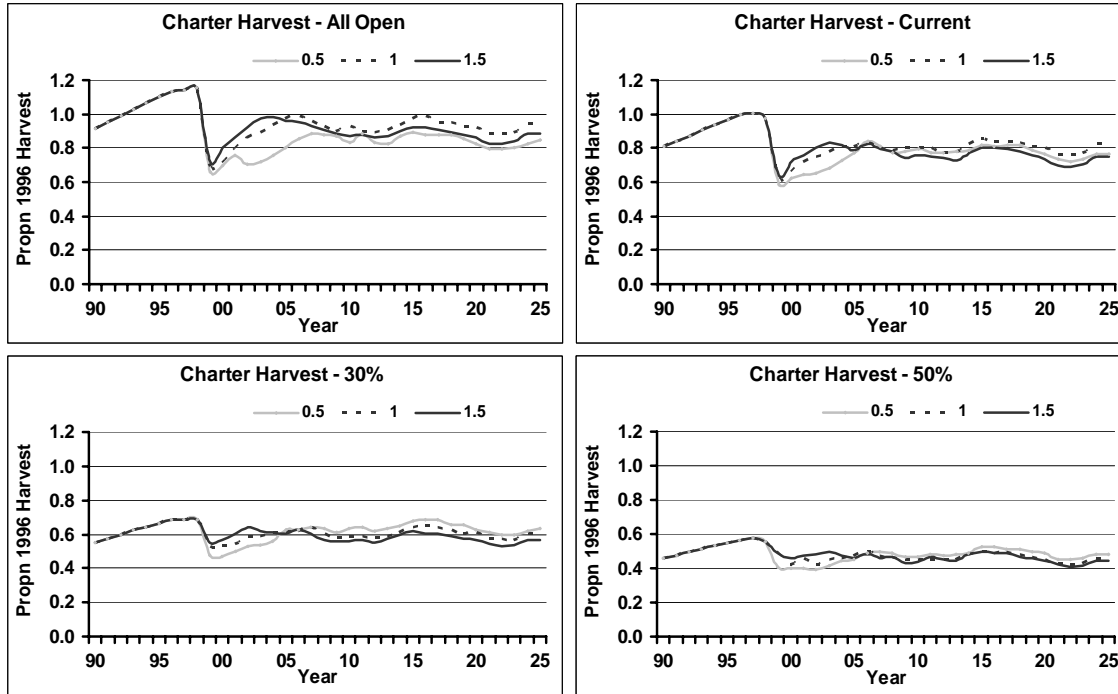


Figure 82: Mean annual recreational catch during the last five years of the projection period (2021-25) relative to the catch in 1998 (Left) and relative to that realised under a 'status quo' projection (Current closures, 1996 effort) (Right) with four effort regimes (**0**, **0.5**, **1.0** and **1.5** x 1996 level) and three closure regimes (current \approx 16%). Values in the right figure are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.



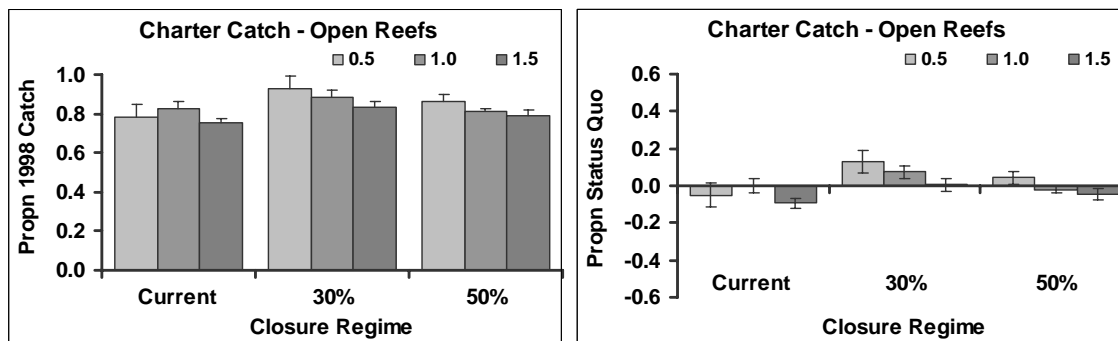
Contrary to results from both commercial and recreational sectors, relative total catch from the charter sector was unresponsive to changes in either effort or closure regime beyond the first 5-10 years of projections (Fig. 83, Fig. 84). By the end of the projection period (2021-2025), harvest by the charter sector actually decreased marginally with increased effort and increased marginally with reduced effort (Fig. 84). The effects of increased closure were more conspicuous for charter catches than for recreational catches, however (Fig. 83 *cf* Fig. 81). With each increase in reef closure, projected harvest by the charter sector dropped by approximately 10-15% of the 1996 harvest, irrespective of effort level (Fig. 83).

Figure 83: Trajectories of charter harvest from open reefs relative to estimated harvest in 1996 from reefs open to fishing then averaged over ten simulations for reefs open to fishing under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5 x 1996 level**). The interval 1996–1998 are years of real data with trajectories thereafter being projections under each strategy set.



Comparison of summaries from the final five years of projections under different management scenarios produced similarly uniform results, with the maximum change from the status quo scenario (1996 effort, current closures) being a 12.6% greater catch under 30% closure and half of the 1996 effort (Fig. 84).

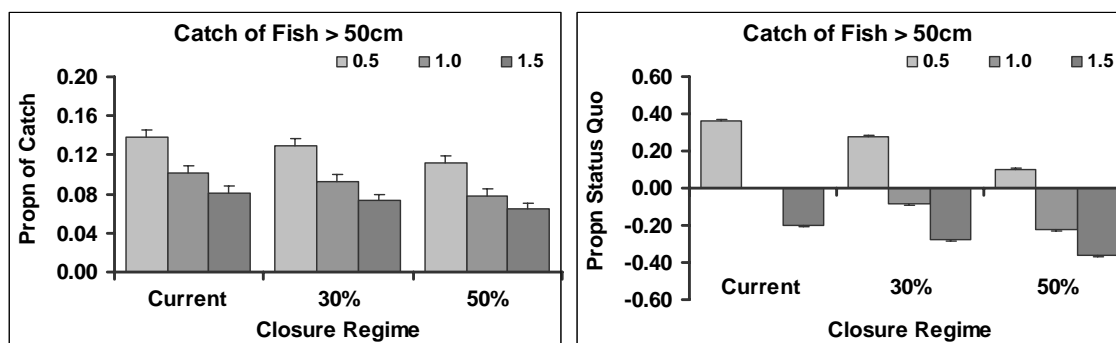
Figure 84: Mean annual charter sector catch during the last five years of the projection period (2021-25) relative to the catch in 1998 (Left) and relative to that from a 'status quo' projection (Current closures, 1996 effort) (Right) from reefs open to fishing under four regimes of fishing effort (**0**, **0.5**, **1.0** and **1.5 x 1996 level**) and three regimes of area closure (Current \approx 16%). Values in the right figure are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.



Proportions of catches comprised of large 'trophy' fish (coral trout > 50cm) varied in direct proportion with effort for each closure regime and were reduced slightly with diminishing fishable areas (increasing closures) (Fig. 85). Proportions of catches represented by large

fish ranged from approximately 6½ -14% (Fig. 85). Compared with status quo projections, recreational fishers would be most satisfied from effort reductions under current closures, with nearly 40% greater likelihood of catching large fish, whilst any increase in overall effort or area closed to fishing would reduce the potential to take large fish compared with current closures under 1996 levels of effort (Fig. 85).

Figure 85: Mean proportion of fish longer than 50cm in catches during the last five years of the projection period (2021-25) (Left) and relative to that realised under a 'status quo' projection (Current closures, 1996 effort) (Right) from reefs open to fishing under four regimes of fishing effort (0, 0.5, 1.0 and 1.5 x 1996 level) and three regimes of area closure (% of reef perimeters; current ≈ 16%). Values in the right figure are scaled so that 0 = no change from 'Status Quo'. Error bars are Standard Errors.



Poached and Discarded Catch

Catch that was under sized and discarded was compared in each year of the projection period with the amount of discard projected under the status quo scenario in the same year (Fig. 86). After an initial period of instability, discards under increased effort scenarios were consistently greater than in all other scenarios and increased slightly throughout time to reach around 1.4 times the reference level over the later years of the projection (Fig. 86). Discards from fishing at 1996 levels under all area closure strategies approximated those under current closures, whilst reducing effort to half of 1996 levels reduced discards by about 50%. Because the projections in Figure 86 are all indexed to a common reference, straightforward comparisons among the plots from different closure regimes are legitimate. Such comparisons indicated that closures reduced discards significantly only when closures were at their greatest (Fig. 86). This result is consistent with the reductions in total harvest under that scenario, predominantly driven by reduced commercial harvest (Fig. 76).

The proportion of the total catch that was under sized in each year decreased with increasing constraints on effort but increased with increasing closure (Fig. 87). The effect of increasing closure was particularly noticeable when closures were increased to 50%, resulting in an increase in percentage of the catch discarded by approximately 20-30% for each effort regime. This effect was less under lower effort regimes than when effort was high (Fig. 87). Thus, the proportion of catch discarded was at a maximum when both effort and closures were at their maxima.

Figure 86: Trajectories of caught and discarded biomass from all sectors compared to that predicted from status the quo scenario (1996 effort, current closures) in the same year averaged over ten simulations for reefs open to fishing under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5** x 1996 level).

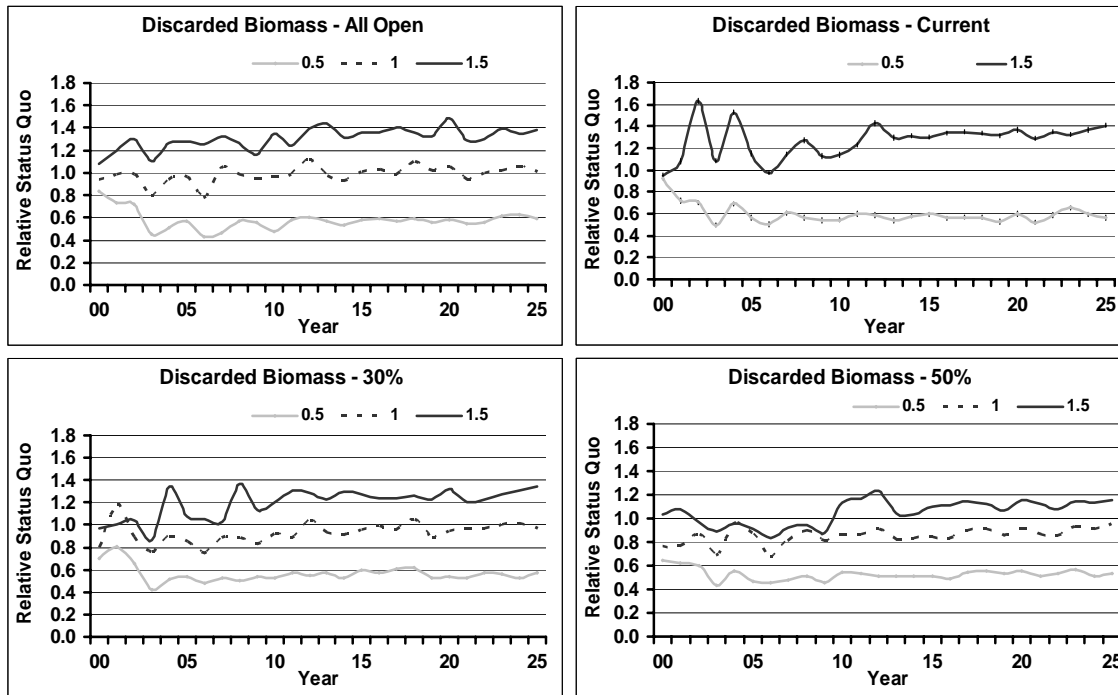
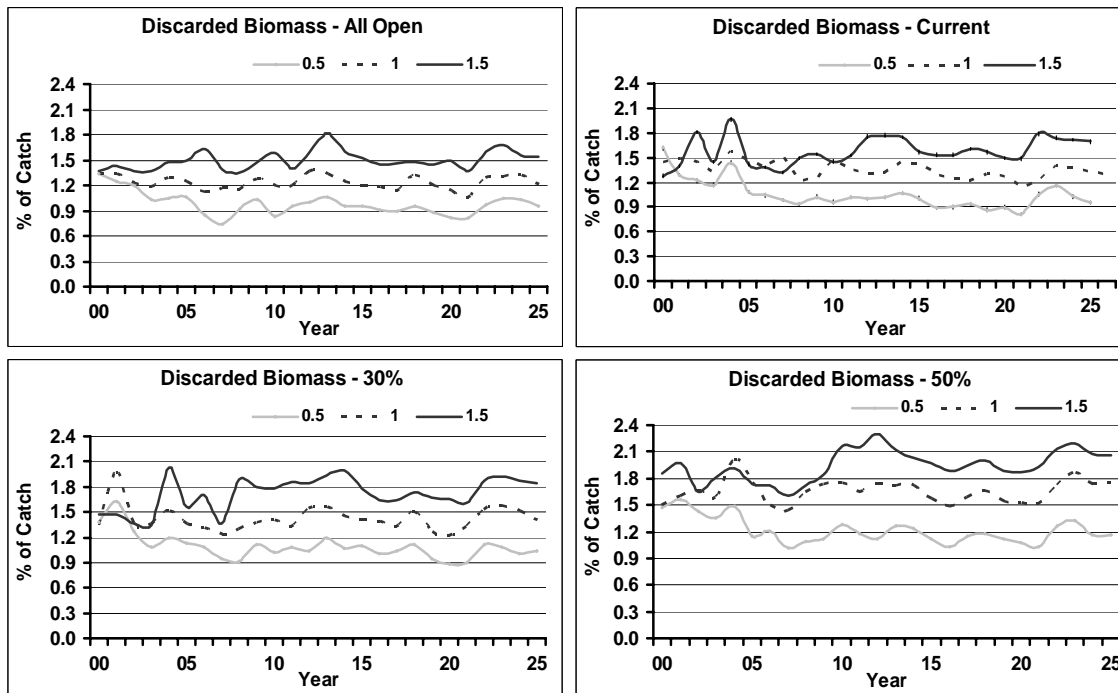
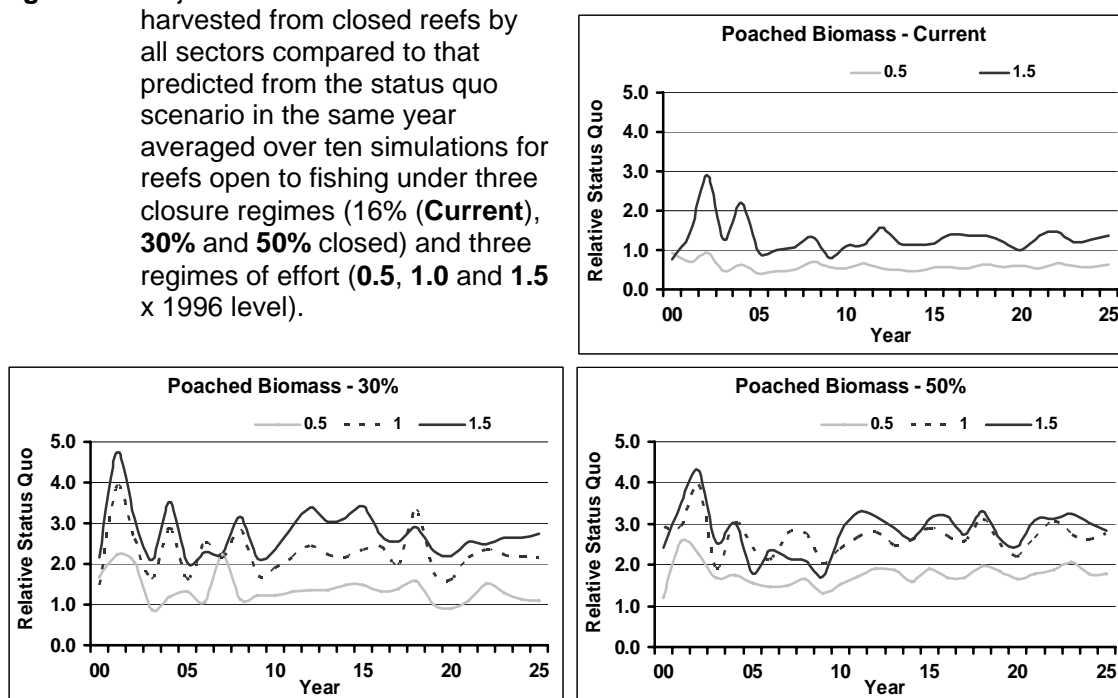


Figure 87: Trajectories of caught and discarded biomass as a percentage of the total catch from all sectors in each year averaged over ten simulations for reefs open to fishing under four closure regimes (**All open**, 16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5** x 1996 level).



Biomass of coral trout harvested by inadvertent or deliberate poaching of closed areas also increased with increasing effort and closure (Fig. 88). This is to be expected because the model assumptions imply that higher levels of effort lead to more fishing effort in closed areas while more closed areas give more opportunity for poaching (*i.e.* more of the biomass is in closed areas). Poached harvest always was least for the reduced effort strategy but differences between 1996 and 1.5 times 1996 effort strategies were generally slight and effectively absent when closures increased to 50% (Fig. 88). Poached harvest increased by more with each increase in closure regime than the proportional increase in amount of closure. For example, poached harvest jumped by 2-3 fold or greater as the perimeter of closures increased by approximately 61% and 148% and the area of closures increased by 73% and 164% from the current levels (Fig. 88).

Figure 88: Trajectories of biomass harvested from closed reefs by all sectors compared to that predicted from the status quo scenario in the same year averaged over ten simulations for reefs open to fishing under three closure regimes (16% (**Current**), **30%** and **50%** closed) and three regimes of effort (**0.5**, **1.0** and **1.5** x 1996 level).

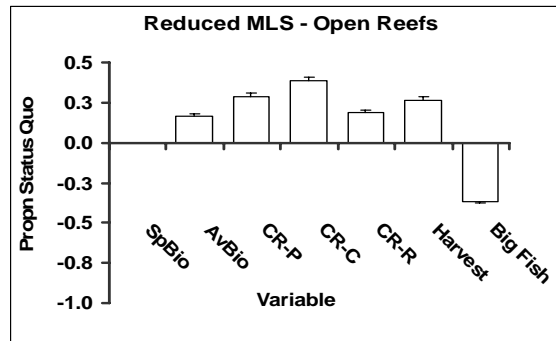


Reduced Minimum Legal Size

Reducing the minimum legal size of harvest during the projection period from the current value of 38cm TL to that at which 50% of coral trout would be expected to be mature (30cm TL) resulted mostly in improved outcomes over status quo projections (Fig. 89). Available biomass, catch rates from all sectors of the fishery, and total harvest were all higher valued at the end of the projection period with lowered MLS than if the current MLS was retained. Differences ranged from 16.5% (available biomass) to nearly 40% (charter fishing catch rates) of the projections with existing MLS (Fig. 89). These increases would all be expected since the proportion of the stock that would be available to the fishery would increase as MLS was reduced. Spawning biomass on closed reefs was effectively unaffected by the change in MLS for harvest. The single 'negative' impact of reducing the legal size for harvest was on the proportion of the catch comprised of fish greater than 50cm TL, which fell by nearly 40% (Fig. 89), in part at least because the total catch increased and the number of small fish in that catch increased. The amount of discarded catch also fell (by 97.5%) under a decreased MLS, largely because the selectivity function for the gear was not changed and

selection of fish smaller than 30cm was low. In reality, a smaller MLS would probably precipitate a shift to smaller hooks, with a commensurate increase in the selectivity of small fish and increased discard rate compared to that we expected with a reduced MLS and no change in selectivity.

Figure 89: Mean spawning biomass on closed reefs (**SpBio**) and available biomass (**AvBio**), catch rates by commercial (**CR-P**), charter (**CR-C**) and recreational (**CR-R**) fishers, total annual harvest (**Harvest**) and proportion of landed coral trout >50m TL (**Big Fish**) from reefs open to fishing during the last five years of the projection period (2021-25) when the minimum legal size for harvest (**MLS**) was reduced to that at which 50% of fish would be mature under 50% closures and 1996 level of effort relative to the analogous values realised with the current MLS under the same closure and effort regime. Values are scaled so that 0 = no change from '50% closed, 1996 Effort, Current MLS'. Error bars are Standard Errors.



Performance Summary

Compiling performance indicators for seven objectives over the nine sets of stakeholder specified management strategies indicated that five of the seven performance indicators would be maximised when effort was reduced under the current closure regime (Table 9). The two exceptions were the conservation performance indicator (status of spawning biomass on closed reefs) and the harvest performance indicator (maximising commercial harvest), the former being satisfied best by increasing the amount of reef closed to fishing and decreasing effort and the latter maximised by allowing effort to increase or remain at 1996 levels under the current closure regime (Table 9). It is noteworthy also that the second-ranked strategy sets for conservation of spawning biomass were current and 30% closures combined with effort reduction. The 'second best' strategy for six of the seven performance indicators was effort reduction combined with 30% closures (Table 9). This strategy mix would be the second worst, however, in terms of maximising total commercial harvest.

The values for the performance indicators should be considered relative to the agreed targets rather than just in absolute value. When compared to those targets, several of the performance indicators were relatively high valued under several management strategy sets, meaning that the targets for some objective statements also were met by several combinations of area closure and effort control (Table 10). In particular, available biomass was likely to remain above the 30% minimum most of the time under all scenarios and spawning biomass on closed reefs was likely to meet the objective target (above 80% virgin spawning biomass 90% of the time) in all scenarios except those with the greatest effort and current or 30% closure (Table 10). Spawning biomass on closed reefs was likely to be maintained above the desired average (90% virgin level), however, only under the maximum closure and minimum effort scenario. The objectives for individual fishers, maintaining catch rates above 80% of reference levels, were likely to be met frequently only with reduced effort, irrespective of closure regime (Table 10).

Table 9: Summary Performance Indicators from the last five years of the projection period (2021-25) for different objectives under nine management strategies comprising three regimes of fishing effort (**0.5**, **1.0** and **1.5** x 1996 level) each combined with three regimes of area closure (% of reef perimeters; current ≈ 16%, 30% and 50%). Larger values indicate better performance of the strategy mix indicated at the left of the table for the objectives listed in column headings. Shaded cells indicate the best strategy set to achieve each objective whilst bold-italicised values indicate the second best alternative strategy set for each objective. **Key:** **SB** – Spawning Biomass; **AB** – Available Biomass; **VSB** – virgin Spawning Biomass; **VAB** – Virgin Available Biomass.

Management Strategies		Objectives and Performance Indicators						
		Conservation	Fishable Stock	Total Harvest	Economic Commercial	Charter	Satisfaction Recreational	Chart and Rec
<i>Closure</i>	<i>Effort</i>	<i>SB_{Closed} : VSB</i>	<i>AB_{Open} : VAB</i>	<i>Catch : Catch₁₉₉₆</i>	<i>CPUE : CPUE₉₃₋₆</i>	<i>CPUE : CPUE₉₆₋₈</i>	<i>CPUE : CPUE₉₈</i>	<i>Fish > 50cm</i>
Current	0.5	0.59	0.96	0.71	1.10	1.34	1.20	0.14
	1.0	0.47	0.82	0.91	0.72	0.71	0.81	0.10
	1.5	0.35	0.63	0.95	0.50	0.43	0.60	0.08
30%	0.5	0.58	0.95	0.66	1.02	1.21	1.16	0.13
	1.0	0.47	0.78	0.82	0.64	0.58	0.78	0.09
	1.5	0.41	0.59	0.84	0.45	0.36	0.58	0.07
50%	0.5	0.63	0.91	0.55	0.86	0.97	1.08	0.11
	1.0	0.56	0.70	0.64	0.50	0.46	0.72	0.08
	1.5	0.52	0.55	0.64	0.34	0.30	0.53	0.06

Table 10: Summary of performance of different strategy mixes relative to target objective statements. All values are the percentage of years from the last five years of the projection period (2021-25) over 10 simulations of each strategy set in which the stated performance indicator met or exceeded the desired target. Larger values indicate better performance of the strategy mix indicated at the left of the table for the objectives listed in column headings. Shaded cells with bold type indicate the strategy set(s) that would achieve each objective. **Key:** **SB** – Spawning Biomass; **AB** – Available Biomass; **VSB** – virgin Spawning Biomass; **VAB** – Virgin Available Biomass; **P()** – Probability of result indicated inside parentheses.

Strategies		Objectives and Performance Indicators					
		Conservation		Stock	Com. CPUE	Chart. CPUE	Rec. CPUE
		$P(SB_{Closed} > 0.8VSB)$	$P(SB_{Closed} > 0.9VSB)$	$P(AB_{Open} > 0.3VAB)$	$P(CPUE > 0.8CPUE_{93-6})$	$P(CPUE > 0.8CPUE_{96-8})$	$P(CPUE > 0.8CPUE_{98})$
Closure	Effort						
Current	0.5	90	72	90	90	90	90
	1.0	90	48	90	0	0	14
	1.5	76	0	90	0	0	0
30%	0.5	90	72	90	90	90	90
	1.0	90	42	90	0	0	4
	1.5	86	0	90	0	0	0
50%	0.5	90	76	90	32	90	90
	1.0	90	70	90	0	0	0
	1.5	90	58	90	0	0	0

Discussion

The comparisons of alternative management strategies we have presented clearly indicate the trade-offs likely to be associated with policy decisions on future management of the Reef Line Fishery on the Great Barrier Reef. The relevance of this work has been highlighted in recent years by the substantial growth in commercial fishing effort (QFS 2002) and the resolution of target amounts of no-take closures for all bioregions on the GBR under the GBRMPA's Representative Areas Program.

The most recent estimates place current commercial fishing effort at or above 1.5 times that in 1996 that we modelled as an upper limit to likely effort growth. Just two years ago, such levels of effort were considered unlikely for the RLF, but it has been suggested by some that effort creep may mean that the *effective* effort is now even greater than the raw number of reported fishing days would suggest. The proposed closure regime for the entire GBR of 25% by area over all bioregions will mean considerable increases in the amounts of 'no-fishing' areas, especially in the southern $\frac{1}{2}$ - $\frac{2}{3}$ of the Marine Park, where fishing effort is traditionally greatest. The 25% target for the RAP perhaps most likely approximates the 30% closure regime we modelled, given the overall increases in closure that will arise as a result of balancing closures of reef-habitats over all bioregions. The results we present above, therefore, provide some clear indications of the consequences of these changes in real circumstances for the objectives most important to various stakeholders.

Our results suggest that the currently elevated level of effort will reduce significantly the prospects of fishers in all sectors realising their objectives in future years, irrespective of the ultimate amount of habitat closed to fishing under the Representative Areas Program. Reducing effort, conversely, is the strategy considered in our Management Strategy Evaluations most likely to realise direct fisheries-related objectives in the areas open to fishing.

Changing effort had relatively little impact, however, on most performance indicators for closed areas, especially conservation of spawning biomass of coral trout. The most effective mechanism by which to increase net spawning stock biomass clearly was to increase the area closed to fishing, presuming that compliance with those closures was relatively high.

Although we did not model explicitly the impacts of varying infringements of closures, our results arise in spite of some infringements because of effort 'leakage' into closed reefs (see methods). The rates of infringement of closed areas that arose in our simulations were approximately 2-4% by total catch per unit of area. That is, catch taken from closed areas was approximately 2-4% of that taken from comparable amounts of open areas at the end of projection periods. We cannot estimate how close these rates are to actual rates of infringement, whether deliberate or accidental. The rates of infringement in our results, however, will have varied in direct proportion to the amount of effort in the vicinity of closed reefs and so would have been considerably higher than the overall GBR average around the high catch reefs and the southern GBR generally. This is likely to be a reasonable representation of reality and the results we have presented are robust to such effects. It is likely, however, that the relative benefits gained from area closures (such as the maintenance of a refuge of near-virgin spawning biomass) will be increasingly compromised by increasing infringement.

It is important to note that the status of coral trout populations in areas open to fishing remained relatively robust under all strategies we considered. For example, even under the most 'adverse' scenario we considered (maximum effort constrained to the smallest fishable area), spawning biomass (in the open areas) remained above 50% of virgin spawning biomass and available biomass remained above 30% of virgin available biomass. These statistics generally would be considered acceptable for a harvested stock managed under conventional input (effort, gear regulation) or output (catch quota) strategies (Hilborn and Walters 1992, Haddon 2001). In large part this is likely to be the consequence of the relatively precautionary minimum legal size limit on harvest of common coral trout and the probable supply of recruits from areas closed to fishing.

If the above is the case, post release mortality or harvesting sub-legal sized fish pose potential threats to our conclusion of relatively robust coral trout stocks. Despite the absence of good estimates of post-release mortality of coral trout, we consider our results likely to be relatively robust to infringements of the MLS limit for several reasons. The level of mortality of sub-legal fish, either through illegal harvest or post-release fatalities, we incorporated (15%) was significantly greater than those estimates that do exist for post-release mortality of reef fish taken in relatively shallow water (5-10%). Further, the growth in trade of live reef fish is likely to reduce rather than exacerbate such mortalities in the future for three reasons. First, fish have to survive capture for one to several weeks to reach the market place. Thus, fish are taken from depths that minimize the risk of embolism, increasingly by gear that reduces injury (e.g., barbless hooks) and handled in ways that maximize survival. All of these things are likely to reduce post-release mortality, though to what level we still do not know. Second, the retention of fish alive, rather than as fillet, makes enforcement of minimum legal size limits more straightforward (but see Mapstone *et al.* in review). Third, the increasing reticence of fishing crew to process fish for sale as frozen product, and the value differential between doing so and selling fish alive, means that the risk of under sized coral trout being sold as fillets of lower valued fish is diminished. Thus, we consider that the rate of incidental mortality of sub-legal catch used in the model is conservative.

The scenarios we considered treated all fishing sectors in the same way. That is, we simulated simultaneous increases or decreases in effort in all sectors. Whilst there is conspicuous evidence of recent increases in effort in the commercial sector, no similar evidence exists for either the recreational or charter sector. Nevertheless, our results were qualitatively similar across all sectors. This is particularly interesting because to a large degree there was only relatively minor overlap in the distributions of fishing effort and catch among the sectors, even though they are allowed to access most of the same areas. The substantial segregation of the sectors was evident in the historical data for charter and commercial fishing and arose from our distribution of recreational fishing. Although we allowed for 'exploratory' fishing by the commercial effort class, our restriction of the charter and recreational effort to those sites where they had fished historically could have perpetuated the separation of effort during the projections. Thus, the projected behaviours in sector-specific performance indicators (e.g., CPUE) were likely to be most influenced by changes in scenarios for that sector. The dispersal of larvae among reefs would provide a mechanism for the indirect effects of depletion by one sector on recruitment to reefs fished by another sector, but we could not estimate such indirect (dispersal mediated) interactions among fleets from the simulations run for this report⁹. Accordingly, our conclusions about the relative merits of alternative management strategy sets should have generality across fishing sectors, although the specific regulatory responses necessary to realise different strategies might vary among sectors. For example, to realise 1996 levels of effort would require significant effort reduction in the commercial sector but, based on current information, might require little immediate action for the charter and recreational sectors.

The consequences of changing closure regimes were negligible for recreational harvests. Given that the closure regimes we considered were focused on corral reefs and not on other habitats in the GBR lagoon or near-shore, the difference between the commercial and recreational sectors almost certainly arose because most of the recreational effort was close to the coast and not in the areas most affected by changing closure regimes. In reality, the effects of closures of near-shore areas are likely to be more influential on the outcomes for recreational fishers.

⁹ It is likely that such indirect effects will be asymmetrical in nature, because of the different amounts of catch taken by the different sectors. For example, the consequences of the commercial harvest on overall spawning biomass, and hence recruitment potential to reefs fished predominantly by the recreational or charter fleets, are likely to be greater than the consequences of either recreational or charter harvests on reefs fished by commercial operators simply because the commercial catch is so much greater than catches by either of the other sectors.

The other main difference among the fishing sectors was that effort levels did not affect the trajectories of total harvest from the charter sector, but did affect those from the commercial and recreational sectors. Basically, reducing charter fishing effort resulted in such dramatically enhanced CPUE that the consequences for total catch of having fewer people fishing were countered by the fact that each fisher was taking dramatically more fish per unit of effort than in other scenarios. In reality, this effect would be expected to be tempered somewhat by bag limits. The fact that we didn't see similar results for the other sectors is likely to be a product of effectively 'restricting' charter fishing effort to the (relatively) small number of reefs from which catch was reported in the available data, combined with the relatively slight overlap in the distribution of effort by commercial and charter sectors.

It is noteworthy that the trajectories of relative CPUE for all sectors dropped substantially over the initial years of projections. Reasons for these declines almost certainly varied among sectors. CPUE in the commercial sector had been declining prior to the projection period and the decline in the first few years of projections was consistent with the extension of this trend in the real data. Changes in the charter sector, however, were at odds with the pattern in the real data, where CPUE was relatively constant over the four years prior to projections. This was caused by an initial transience in the effort allocation model in shifting from the historical period, in which catches are dictated by the data, to the projection period in which catches depend on where effort is allocated. Effort allocation in the projection period is based on the principle of maximising CPUE, conditioned on past experiences. Since CPUE is an indicator of abundance, it is apparent in the first year of projection that charter experiences constrained the effort to reefs with low available biomass. Consequently, it is certainly possible that the objective of maximising CPUE does not apply to the charter fishers, and that perhaps they allocate effort according to a different criterion. With only a single year of reference data, it is not clear what prior trends were present in the recreational fishery, but initial transience in the recreational CPUE was also the likely cause for the initial decline of CPUE in the projection period. Because of the lack of data, recreational effort was allocated considerably different from the other fleets, by implementing a static rather than a dynamic effort allocation algorithm (Appendix B).

Another likely origin of these apparently anomalous drops in CPUE for the charter and recreational sectors lies in the resolution of the data from which we estimated prior CPUE. In particular, the resolution of catch and effort data available to us was generally poor and it is likely that the catches of common coral trout, which was what we modelled, were significantly over-estimated by the 'real' catch data we used. This was likely to have occurred for two reasons. First, it was unclear whether reported catch was kept or discarded and so we treated all reported catch as harvest. Second, we erred on the side of inclusion rather than exclusion of catch that was possibly (rather than definitely) coral trout and we counted all catches of 'coral trout' as *P. leopardus*, when in many instances (particularly for near-shore catches) the catch was likely to be *P. maculatus*. Third, the recreational catch data we received were numbers of fish and our conversion to biomass may have over-estimated the biomass of recreational (and charter) catch, especially if we had included as harvest 'catch' that was under sized (and released). These uncertainties, however, would have affected the values of relative CPUE within each scenario but would not have affected the qualitative relationships among management strategy scenarios, which are the primary focus of Management Strategy Evaluations.

It might be argued that several of the assumptions (e.g., the values we used for Habitat Scalar and steepness of the stock-recruitment relationship) and level of aggregation of results (the entire GBR) might have influenced substantially the nature of the results we present. Sensitivity analyses of the simulations support such contentions, however, only in respect of the magnitude of effects of alternative scenarios and the magnitude of contrasts among them. Importantly, the qualitative relationships among scenarios were robust to changes in model parameters. Accordingly, the conclusions about the *relative* merits of increasing or decreasing fishing effort or area closures are robust to most changes in model assumptions.

Importantly, the increases in fishing effort in recent years are most likely to impact most negatively on the performance indicators for areas open to fishing. The likely increase in area closures under the Representative Areas Program is likely to exacerbate the decline in fishery performance, but our results suggest that the growth in fishing effort is likely to be considerably more influential than changes in areas available to the fishery. Given the growth in commercial effort since the inception of this work, there would appear to be substantial economic incentive to the commercial sector and the broader economy to develop specific strategies to reduce total real effort in the commercial sector from current levels. The conundrum in these results, however, is that the improved prospects we suggest from effort reduction would apply only to those fishers remaining with economically viable amounts of effort in the fishery. We are unable to assess the magnitude of costs likely to be incurred by those fishers excluded or severely constrained through the effort reductions that would now be necessary to achieve the two lower effort scenarios we considered.

It should be noted that our evaluations related only to objectives for and about the RLF and the status of the primary target species, common coral trout. We did not consider and make no comment here on the several other objectives, such as conservation of biodiversity and ecosystem function, that are typically associated with the use of Marine Protected Areas as a management strategy. The relative merits of area closures and effort control strategies might look very different if judged against such other sets of objectives. The development of MSE methods to evaluate the performance of alternative fisheries and conservation management strategies against multi-species, ecosystem and broader social and economic objectives remains an important area for further work and one that is on urgent need of attention.

Nevertheless, this research has laid bare some of the inevitable trade-offs among different scenarios for managing the Reef Line Fishery in the Great Barrier Reef World Heritage Area. Most importantly, the trade-offs have been assessed in relation to objectives and performance indicators specified by diverse stakeholders in the fishery and the World Heritage Area. We present the tradeoffs in ways that allow direct comparisons among disparate objectives, essentially providing a common currency for comparing performance across fundamentally different types of objectives. In so doing, we hope that the costs and benefits of different management options are more transparent to all stakeholders than might otherwise have been the case. We hope that such transparency aids in the negotiation of acceptable future management arrangements for the Great Barrier Reef World Heritage Area and the Reef Line Fishery.

Benefits

All stakeholders in the Reef Line Fishery and the Great Barrier Reef Marine Park and World Heritage Area will benefit from empirical insights into the direct and indirect effects of line fishing on the Great Barrier Reef. The results of this work demonstrated a considerable measure of effectiveness in existing management strategies for both the fishery and conservation of the resource. The provision of quantitative assessments of the prospects for alternative management strategies to achieve management and stakeholder objectives provides clear indications of the trade-offs between the costs of various regulatory measures and benefits to the resource and the fishery of those measures. The development of clearly articulated and quantified objectives for the diverse stakeholders in the fishery provides a clearer foundation for negotiation among stakeholders about the strategies that will best meet their respective needs. The provision in a common currency for assessments of the likelihood that objectives would be achieved by one or other management strategies provides a transparent basis for comparing costs and benefits to different stakeholders whose needs are expressed in qualitatively different terms. This information will benefit both the Queensland Fisheries Service (in managing the fishery), the Great Barrier Reef Marine Park Authority (in managing the Great Barrier Reef Marine Park, where most of the reef line fishing occurs), the fishers who seek to retain acceptable access to and benefit from the fishery, policy makers who are required to set future directions for ecologically sustainable use of the GBR World Heritage Area, and conservation lobbyists who seek reassurance that management of the fishery is consistent with biodiversity conservation.

Further Development

There are several areas for further development from the work presented here. In particular, we recommend the following, in addition to work that is underway already:

- Verification of the extent of habitat and the best way to relate biomass to habitat;
- Verification of catchability uncertainties, especially whether catchability declines for large fish?;
- Better resolution of infringement rates;
- Exploration of potential indirect (larval subsidy) benefits of closure;
- Development of models with which to explore indirect effects of fishing, especially those potential effects that might relate to ecosystem objectives of stakeholders;
- Exploration of interactions among different sector-specific effort dynamics;
- Verification of the relationship between Underwater Visual Survey and Catch Survey indices of abundance;
- Exploration of the extent to which persistent regional patterns in abundance might be the product of larval mixing relationships over large distances on the GBR;
- Exploration of interactions between small and large fish around baits;
- Exploration of the degree to which the ELFSim models might be simplified, and made more efficient, without significant loss of predictive power;
- Development of explicit feed-back links between the effort allocation and management components of the ELFSim package.

Conclusion

We have shown that reef line fishing has significant impacts on the primary target stocks in some regions of the Great Barrier Reef. Fish in areas available to the fishery were less abundant, smaller and younger than in areas historically protected from fishing. Manipulations of fishing effort and protection status showed that these differences most likely reflected the impacts of fishing in harvested areas and the protection from impacts of fishing in the closed areas, rather than pre-existing differences between areas selected for closure or left open to the fishery. The magnitude of these impacts varied approximately in proportion to the amount of fishing effort. Deliberate or inadvertent infringement of closures and any migration of fish across closure boundaries have been insufficient so far to counter completely the effects of protecting local populations from harvest. Thus, areas closed to fishing by the Great Barrier Reef Marine Park Authority via zoning have been effective in ameliorating the impacts of fishing and protecting significant portions of the coral trout and red throat emperor populations on the GBR where fishing effort has been greatest.

There was little convincing evidence of consistent secondary effects of fishing on, for example, the prey of the primary harvest species, common coral trout. The absence of such effects might arise because of confounding influences of significant regional variations in abundances of both predators and prey and the amount of fishing mortality on the predators, as well as apparently large inter-annual changes in prey abundances which were probably driven by recruitment variation.

Evaluations of alternative management strategies posited by a diversity of stakeholders showed significant differences in the outcomes to be expected from the alternatives. Most stakeholder objectives were likely to be met most effectively by control of fishing effort. Area closures were effective in preserving the mature component of populations of coral trout in spite of minor levels of infringement. The utility of the area closures was most compromised when fishing effort was greatest, and the effects of increased fishing effort were most severe when area closures were greatest.

Despite the clear evidence of trade-offs between closure and effort regulation strategies and clear impacts of fishing, the simulations indicated that under all strategy sets, populations of common coral trout were likely to remain biologically robust to harvest. This projection was most likely the product of the currently generous minimum legal size of harvest, even in the absence of area closures. The inference was robust to moderate levels of mortality of sub-legal sized fish.

Importantly, the increases in fishing effort in recent years are most likely to impact most negatively on the performance indicators for areas open to fishing. The likely increase in area closures under the Representative Areas Program is likely to exacerbate the decline in fishery performance, but our results suggest that the growth in fishing effort is likely to be considerably more influential than changes in areas available to the fishery.

Finally, it should be noted that our results to date directly address only the impacts of harvest on the common coral trout, *P. leopardus*, and cannot be taken to indicate how other species exploited by the Reef Line Fishery might respond to sustained harvest. Limited information about the diversity among species in abundance, life-history, distribution and population dynamics are likely to mean that many species are likely to be less robust to harvest than the common coral trout. Accordingly, conservative regulations for the harvest of those species would seem prudent at this stage.

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Appendix A: Applying a Population Dynamics Model to ELF Experiment Data

Introduction

One of the objectives of the ELF experiment is to obtain data from which to estimate some of the quantities to parameterise operating models for common coral trout (*P. leopardus*) on the Great Barrier Reef. These operating models form the basis for the evaluation of alternative management strategies in terms of their ability to satisfy the agreed management objectives for the Great Barrier Reef and the Reef Line Fishery (Mapstone *et al.* 1998, Davies *et al.* 1998).

Punt *et al.* (2001) used Monte Carlo simulation techniques to evaluate the ability of a variety of potential methods for estimating quantities that could be used to parameterise operating models. Methods (estimators) were evaluated in terms of their ability to estimate fishable biomass (relative and absolute levels), exploitation rates, the instantaneous rate of natural mortality (M), and the relationship between commercial catch rates and abundance. None of the estimators examined by Punt *et al.* (2001) were capable of establishing the relationship between commercial catch rates and abundance reliably. However, an estimator based on the Deriso (1980) delay-difference model that used the Underwater Visual Survey (UVS), the research line fishing survey, and the mean age data estimated the remaining quantities reasonably successfully by the end of ELF experiment if the pulsing of reefs was achieved and the data were collected with the sample sizes and frequency assumed during the simulations. The ability to estimate biomass and fishing mortality was severely compromised when only the data up to 2000 were used.

Annex A1 provides the technical details of a slightly modified version of the estimator selected by Punt *et al.* (2001). The modifications involve including a correction factor derived by Schnute (1985) to the basic dynamics equation to account for the fact that the mass of an animal one year before recruitment is non-zero (as is assumed by the Deriso delay-difference model) and assuming that the mean age data are absolute rather than relative indices of the actual mean age of the catch (i.e. the value of the parameter ϕ in Equation A.5 is assumed to be 1 rather than being estimated from the data). The first modification was made because the original formulation led to unrealistic estimates for the mean weight of animals aged 5 and older (20kg+) while the second modification was made because the estimate of the bias between the actual and the observed mean age of the catch was almost 50% - a value considered unrealistic and completely inconsistent with the levels for this bias underlying the simulations on which the estimator was based. The impact of using the original rather than the modified estimator is considered in one of the tests of sensitivity.

This appendix applies variants of the estimator in Annex A1 to the ELF experimental data available up to 2000¹. The purpose of this application is to assess how sensitive key outputs are to different formulations of the estimator and whether the assumptions underlying the simulations upon which the estimator was based remain valid given the data so far collected.

The Data and Derived Inferences

A summary of the ELF experimental data upon which the analyses of this appendix are based is provided in Table A1. Overall, the data have been collected at the expected frequency and with age-structure sample sizes not too different from those assumed by Punt *et al.* (2001). The major differences between the data available for estimation purposes and the assumptions implicit in the simulations considered by Punt *et al.* (2001) relate to the sample sizes for 1995 (the actual sample sizes are smaller) and the availability of the age-composition data for 2000 and for the Autumn and Winter surveys during 1997 and 1999.

¹ Some information is available for 2001 but was ignored for the analyses in this appendix.

Table A1: Overview of the data available for inclusion in the analyses. a) - Key to reef names and treatments represented by abbreviations in b)-d); b) - Underwater Visual Survey data; c) - Research line fishing surveys; d) - Age-composition sample sizes (all ages). In b) and c) entries of '1' indicate data available, '0' indicate data not available.

Table A1-a): Key to reef abbreviations.

Lizard Region			Townsville Region		
Abbrev	Reef Name	Treatment	Abbrev	Reef Name	Treatment
L1	MacGillivray	MNP-Control	T1	Dip	MNP-Control
L2	South Direction	MNP-Control	T2	Glow	MNP-Control
L3	Rocky-A	MNP-Fished (1)	T3	Yankee	MNP-Fished (1)
L4	Eyrie	MNP-Fished (2)	T4	Faraday	MNP-Fished (2)
L5	Rocky-B	GU-Fished (1)	T5	Knife	GU-Fished (1)
L6	14-133	GU-Fished (1)	T6	Fork	GU-Fished (1)

Mackay Region			Storm Cay Region		
Abbrev	Reef Name	Treatment	Abbrev	Reef Name	Treatment
M1	20-137	MNP-Control	S1	21-131	MNP-Control
M2	20-142	MNP-Control	S2	21-132	MNP-Control
M3	20-136	MNP-Fished (1)	S3	21-130	MNP-Fished (1)
M4	Bax	MNP-Fished (2)	S4	21-133	MNP-Fished (2)
M5	Liff	GU-Fished (1)	S5	21-134	GU-Fished (1)
M6	Boulton	GU-Fished (1)	S6	21-139	GU-Fished (1)

Table A1-b): Availability of Underwater Visual Survey Data (1=Yes; 0=No)

Year	Season	Region and Reef																							
		Lizard Region						Townsville Region						Mackay Region						Storm Cay Region					
		L1	L2	L3	L4	L5	L6	T1	T2	T3	T4	T5	T6	M1	M2	M3	M4	M5	M6	S1	S2	S3	S4	S5	S6
1995	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1996	Sum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1997	Summer	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Winter	0	0	1	1	1	0	0	0	1	1	1	0	0	0	1	1	1	0	0	0	1	1	1	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1998	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1999	Summer	0	1	0	1	0	1	0	0	0	0	0	1	1	0	0	1	0	1	1	0	0	1	0	1
	Autumn	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Winter	0	1	0	1	0	1	1	0	0	1	0	1	1	0	0	1	0	1	1	0	0	1	0	1
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
2000	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Table A1-c): Availability of Research Line Fishing Catch Survey Data (1=Yes; 0=No)

Year	Season	Region and Reef																							
		Lizard Region						Townsville Region						Mackay Region						Storm Cay Region					
		L1	L2	L3	L4	L5	L6	T1	T2	T3	T4	T5	T6	M1	M2	M3	M4	M5	M6	S1	S2	S3	S4	S5	S6
1995	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1996	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1997	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	1	0	1	1	0	1	1	0	1	0	0	1	0	1	0	1	1	1	0	1	0	0
	Winter	0	0	1	0	1	1	0	1	1	0	1	0	0	1	0	1	0	1	1	1	0	1	0	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1998	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	1	0	1	1	0	1	1	0	1	0	0	1	0	1	0	1	1	1	0	1	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1999	Summer	0	1	0	1	1	0	0	0	1	1	0	1	1	1	0	0	1	0	0	0	0	0	0	0
	Autumn	0	1	0	1	1	0	0	0	1	1	0	1	1	1	0	0	1	0	0	1	1	0	0	1
	Winter	0	1	0	1	1	0	0	0	1	1	0	1	1	1	0	0	1	0	0	1	1	0	0	1
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
2000	Summer	0	1	0	1	1	0	0	0	1	1	0	1	1	1	0	0	1	0	0	1	1	0	0	1
	Autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Table A1-d): Sample Sizes of Age Composition Data (Number of otoliths reliably aged)

Year	Season	Region and Reef																							
		Lizard Region						Townsville Region						Mackay Region						Storm Cay Region					
		L1	L2	L3	L4	L5	L6	T1	T2	T3	T4	T5	T6	M1	M2	M3	M4	M5	M6	S1	S2	S3	S4	S5	S6
1995	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	43	130	160	88	147	128	88	180	110	48	46	59	298	233	308	190	101	201	278	297	296	170	166	158
1996	Sum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	0	46	67	145	69	72	42	144	106	32	58	56	161	242	258	217	103	89	228	263	265	142	101	146
1997	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	72	56	171	123	105	106	121	154	15	120	24	66	73	128	111	104	74	33	202	134	101	89	139	64
1998	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	1	0	1	0	0	0	0	0	0	0	0	0	2	1	1	0	0	0	0	1	3	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	70	116	102	142	76	99	160	169	83	110	51	87	151	193	181	238	89	129	270	134	151	107	149	89
1999	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	49	126	167	99	140	99	103	218	151	92	96	65	129	212	237	174	187	134	273	263	256	137	211	172
2000	Summer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Spring	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Average		47	95	96	100	90	101	86	173	93	67	55	67	162	202	183	154	79	117	313	273	214	129	110	157

Figure A1 plots the coefficient of variation, CV, of the UVS and research line fishing indices against these indices for each of the 24 experimental reefs. The assumption underlying the estimator and the simulations of Punt *et al.* (2001) is that the CVs would be independent of the indices and equal to 0.2 (the dotted line in Figure A1). The results in Figure A1 suggest that this assumption is not violated to a substantial extent.

Catch data are available from the QFS at the level of 6'x6' site². The base-case rule used to allocate catches (by site) to reefs for these analyses is as follows. If a reef is closed, the catch from that reef is assumed to be zero and the reef is then ignored for the purposes of catch allocation. If more than one fished reef occurs within a site, the catches are allocated to reefs in proportion to reef perimeter. Catch data are not available for 2000, but this should not be important because all experimental reefs were closed to fishing during most of 2000.

The simulations of Punt *et al.* (2001) assumed that the catches from closed reefs would not be available for use by the estimator even if infringement of closures occurred. In contrast, information on catches from reefs nominally closed are available for the entire 1995-99 period (see, for example Figure A2 which plots the catches for the closed and pulsed reefs in the northernmost, Lizard, region). The bulk of the analyses ignore catches from closed reefs, essentially assuming that such catches are reporting errors or are from open reefs in the same reporting site as the closed reefs concerned. In reality, however, the possibility of fairly substantial poaching cannot be ruled out as a plausible hypothesis. Punt *et al.* (2001) show that poaching from closed reefs will lead to possibly severe reductions in the accuracy and precision of the results from any analyses of the data from the ELF experiment.

² Although some data are provided only by 30' x 30' grids, all data are disaggregated to 6' x 6' sites prior to analyses. See methods sections of the report for details of how the disaggregation is done.

Figure A1-a): Coefficients of variation of the UVS indices *versus* these indices. The dotted line indicates the coefficient of variation assumed by the estimator and by the simulations on which it was based.

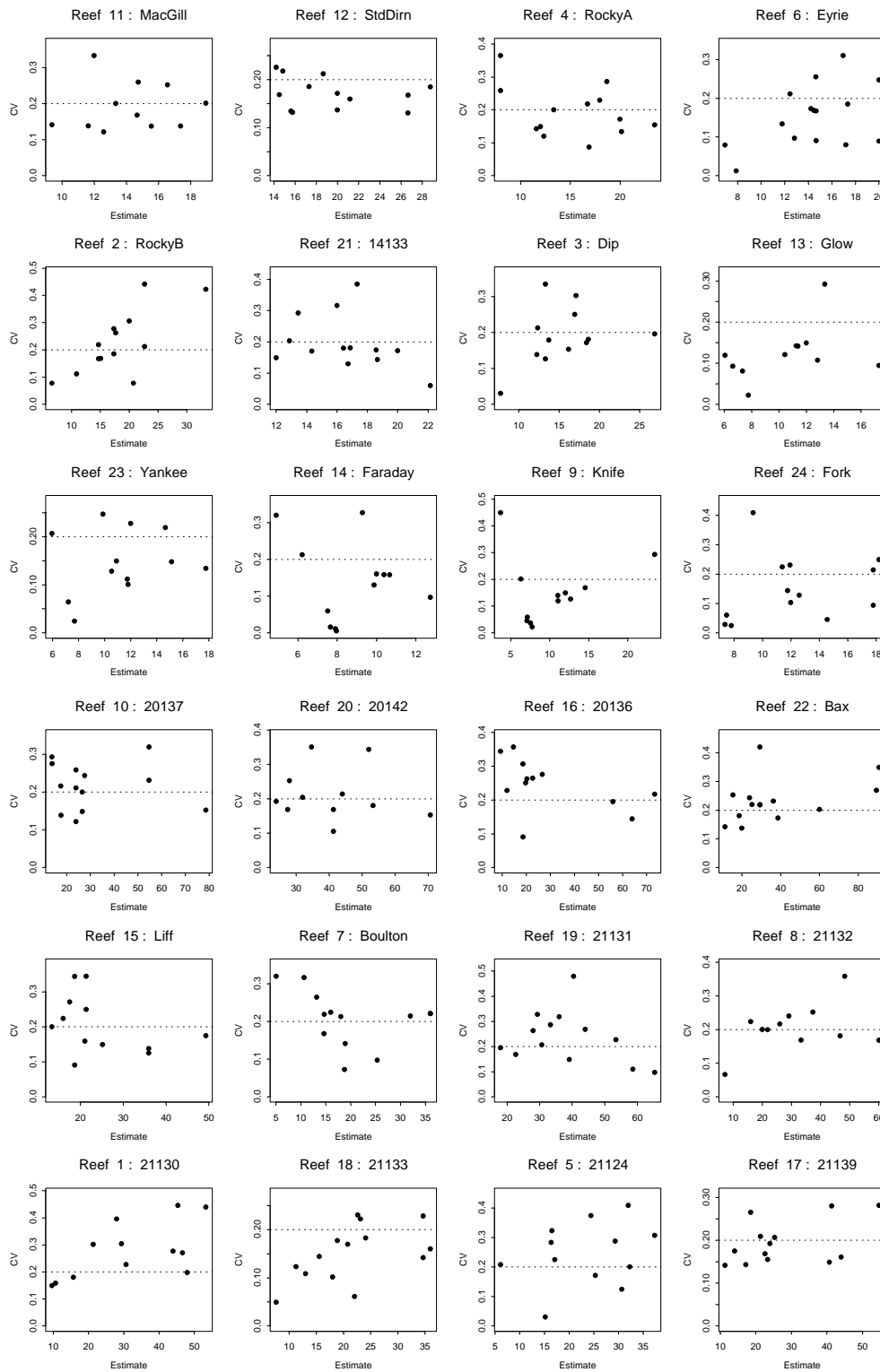


Figure A1-b): Coefficients of variation of the catch line fishing indices *versus* these indices. The dotted line indicates the coefficient of variation assumed by the estimator and by the simulations on which it was based.

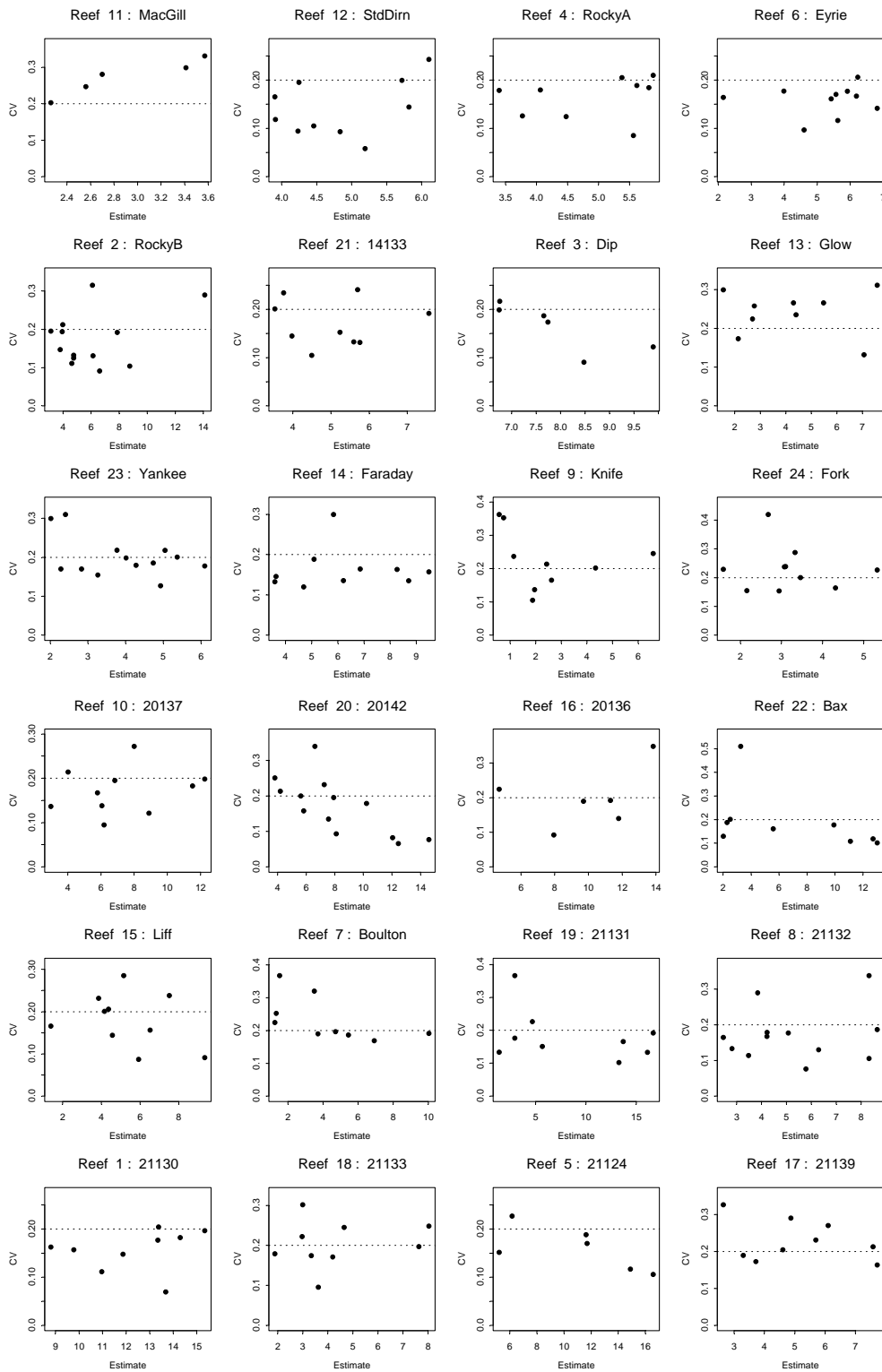
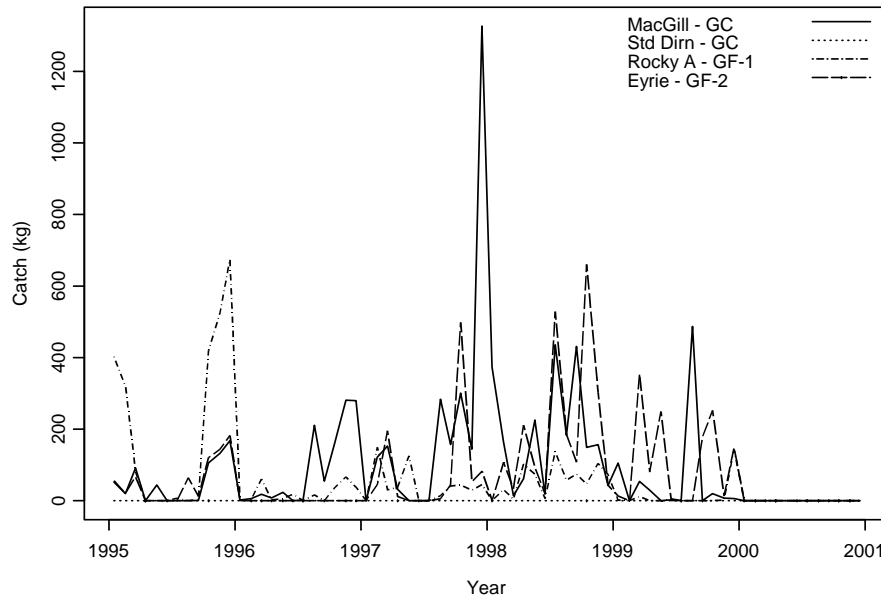


Figure A2: Catches recorded for the MNP-Control and MNP-Fished reefs in the Lizard region. The catches from April 1997 – March 1998 (Rocky A) and March 1999 – February 2000 (Eyrie) can be attributed to the attempt to pulse the “MNP-Fished” reefs.



Fitting the Model to the Data

Base-case analysis

Figure A3 shows the time-trajectories of exploitable (4+) biomass in tonnes during 1995-2001 for each of the 24 experimental reefs for the base-case analysis (Table A2 lists specifications for this analysis and seven sensitivity tests). Figure A4 shows the fits to the UVS, research line fishing, and mean age indices over the same period. Figure A5 examines the fits to the three data sources further by plotting observed and model-predicted indices.

Figure A3: Estimated exploitable biomass trajectories for 24 reefs in the ELF Experiment.

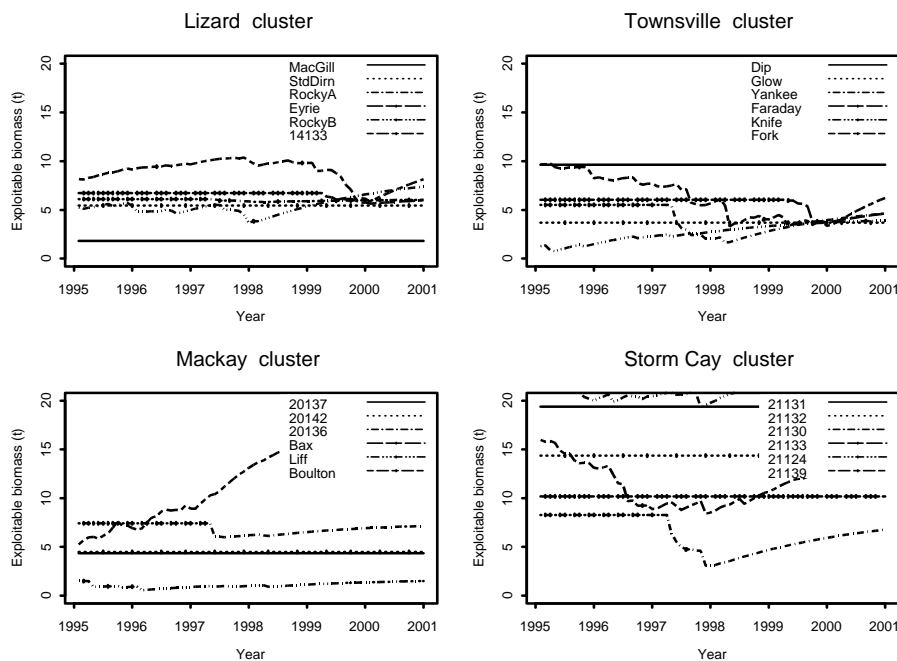


Figure A4-a): Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Lizard region.

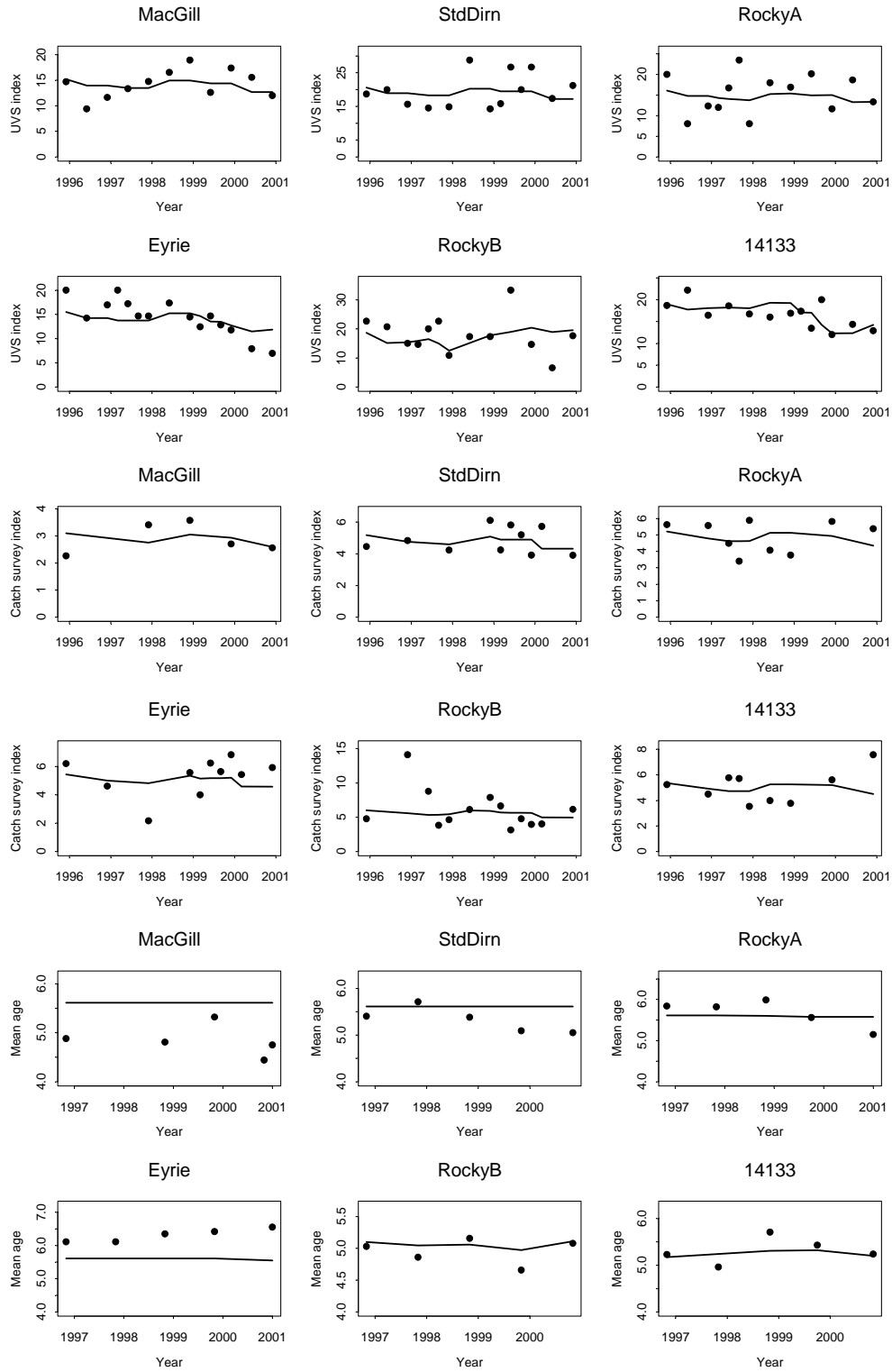


Figure A4-b): Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Townsville region.

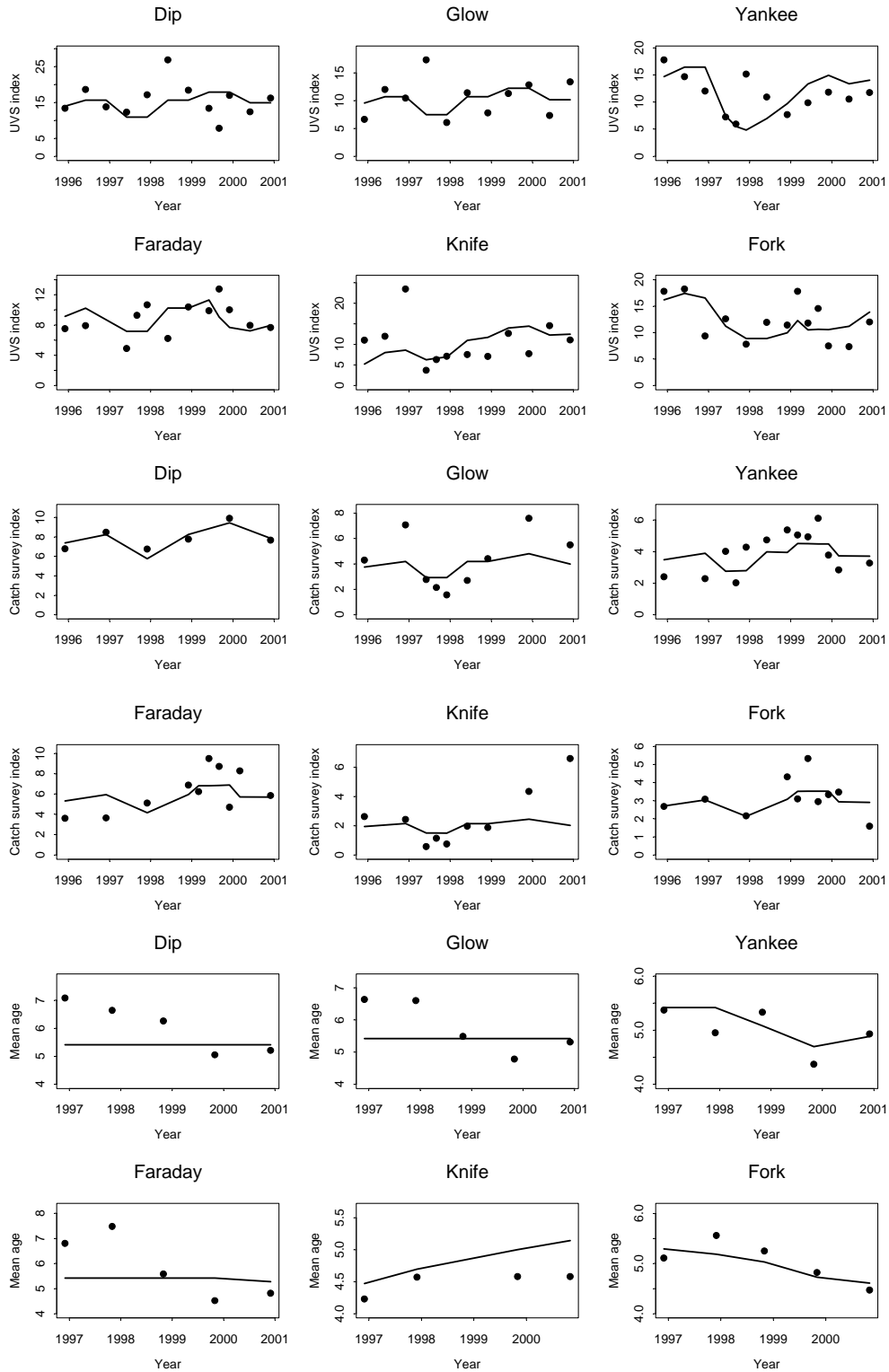


Figure A4-c): Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Mackay region.

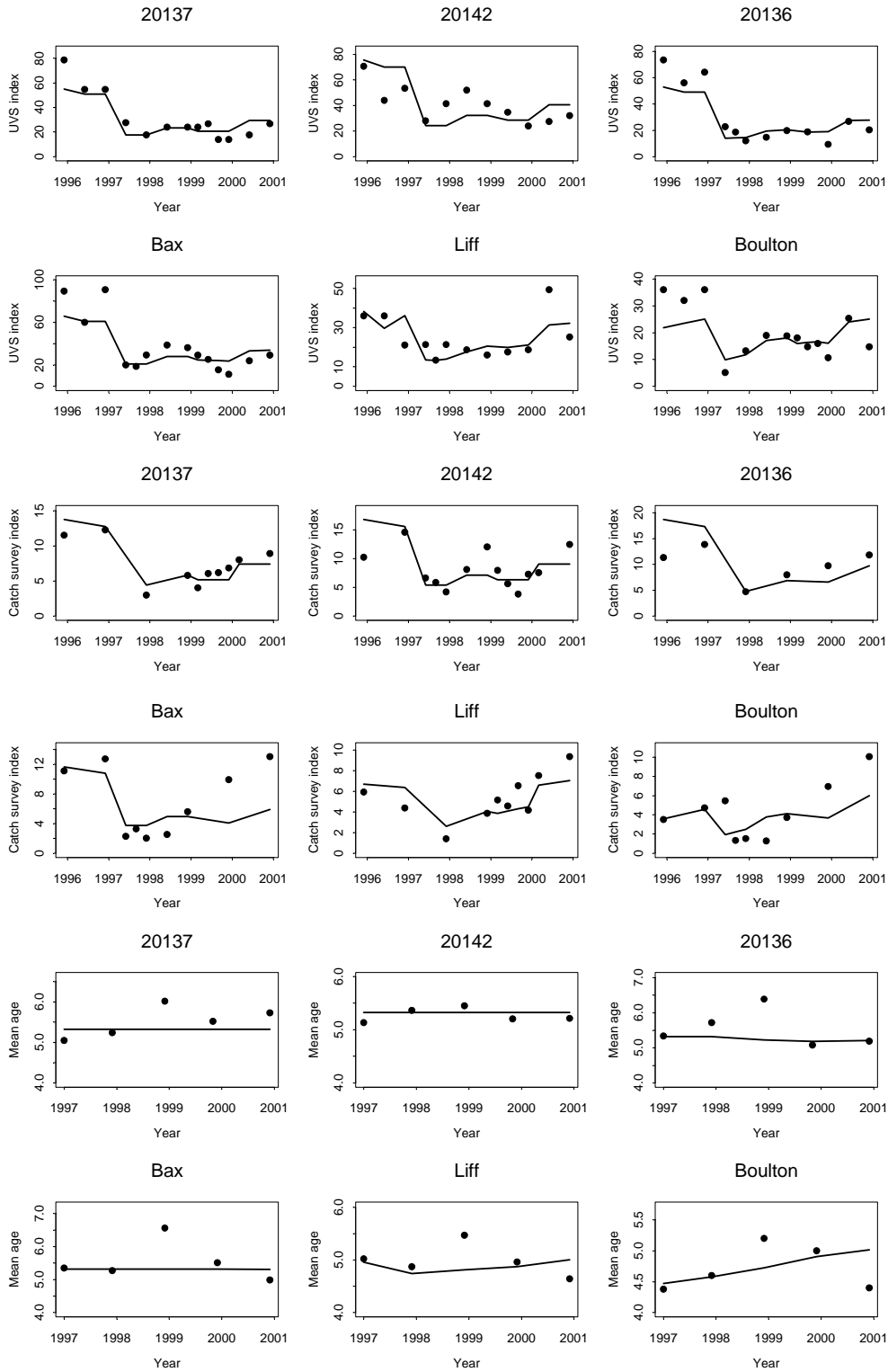


Figure A4-d): Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Storm Cay region.

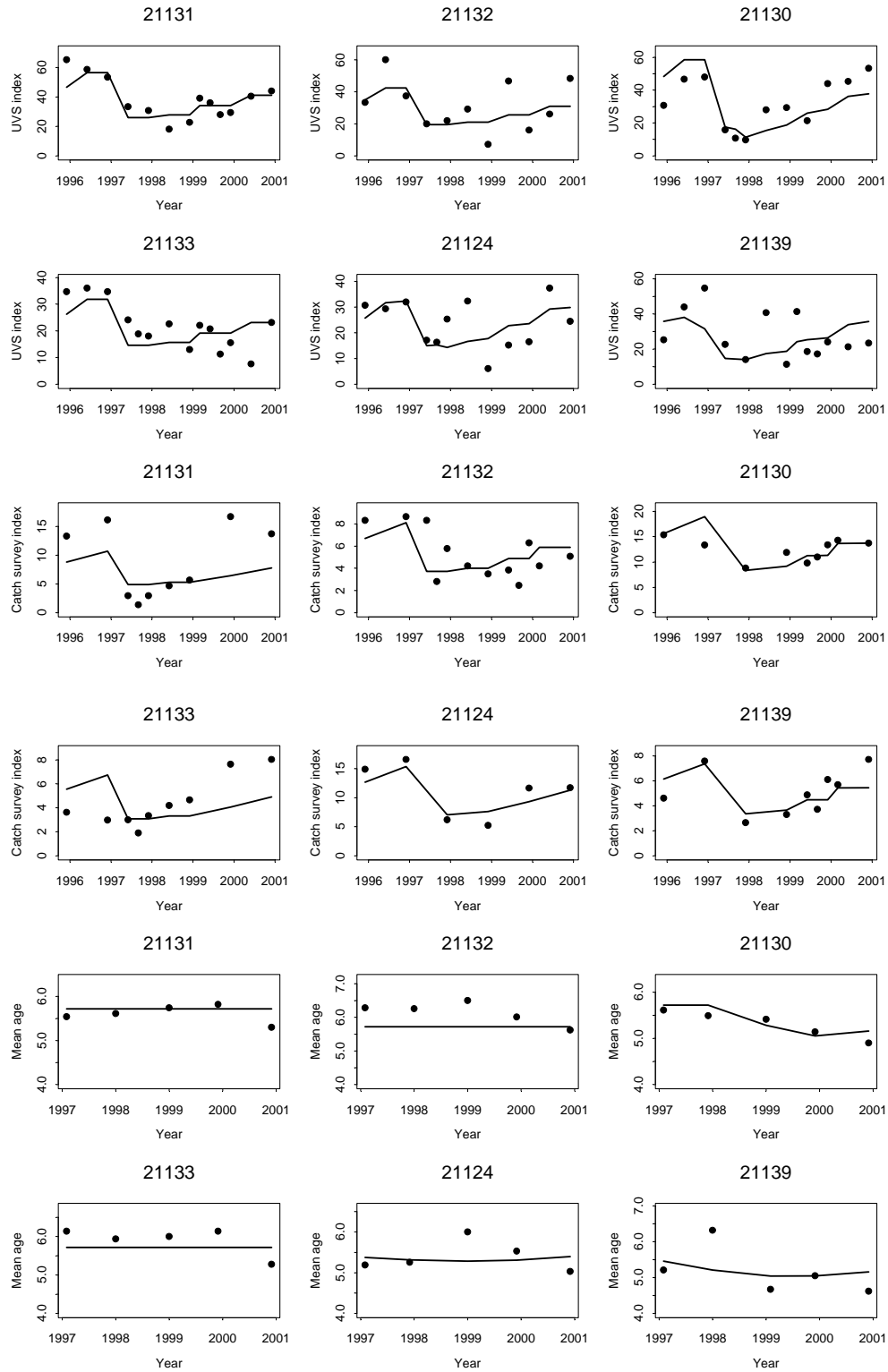


Figure A5-a): Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Lizard region.

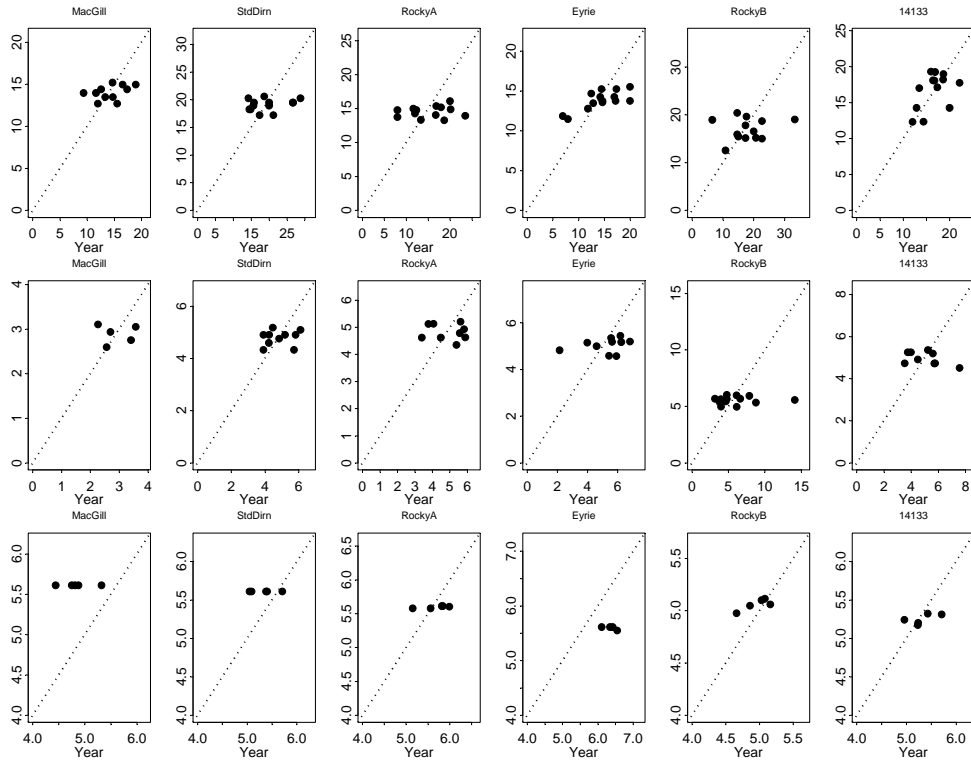


Figure A5-b) : Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Townsville region.

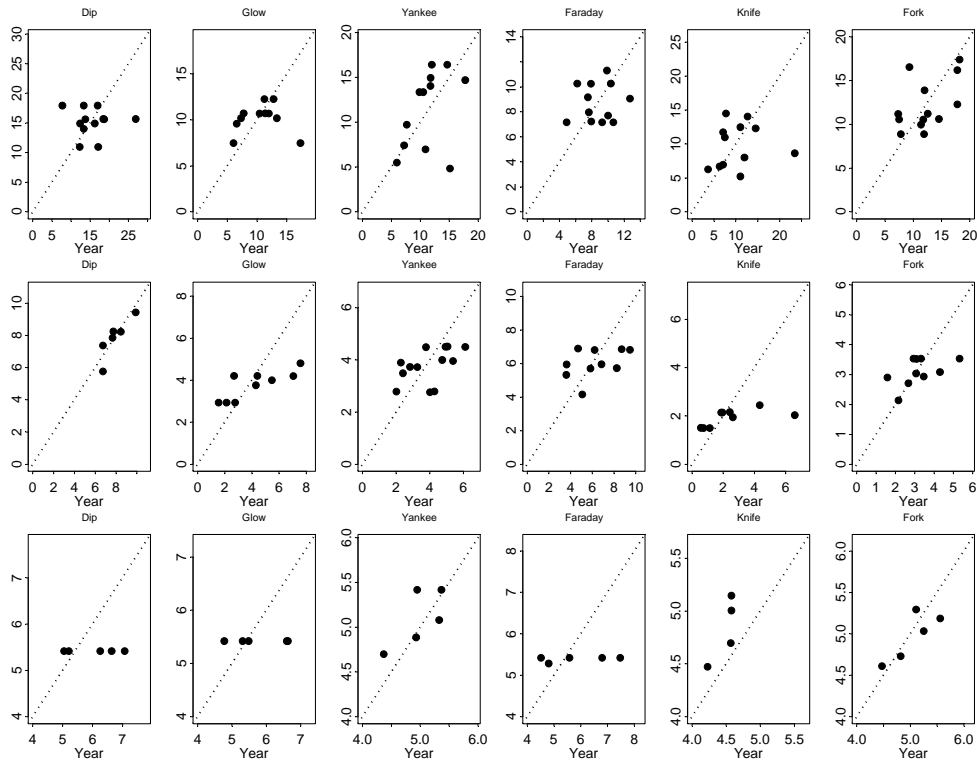


Figure A5-c): Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Mackay region.

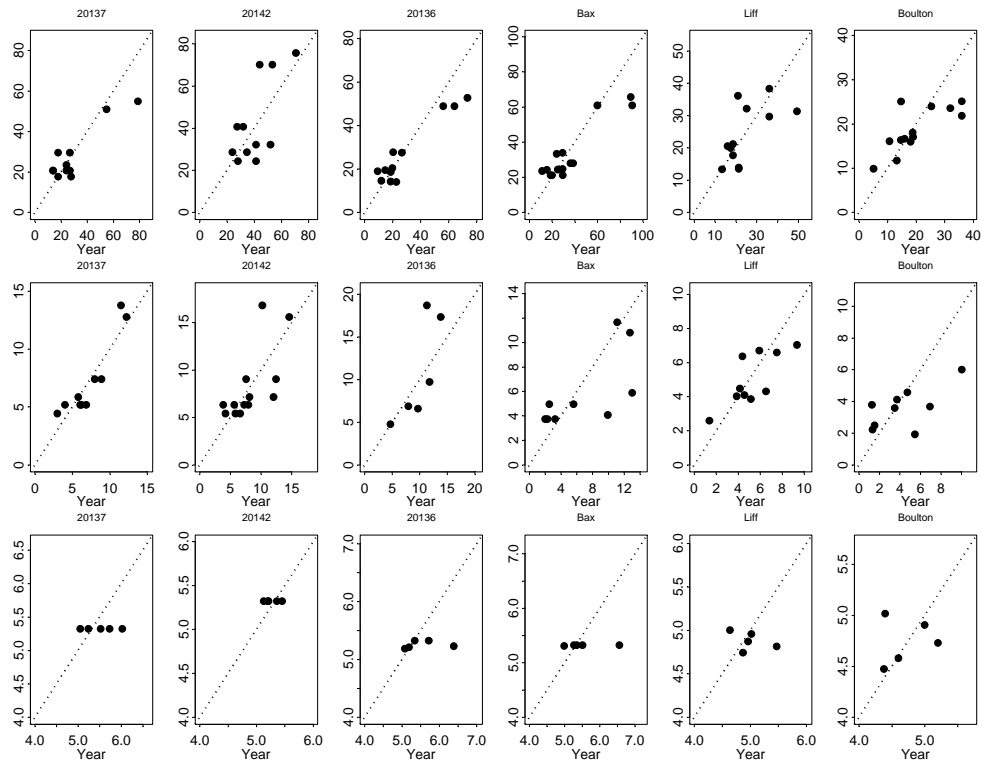


Figure A5-d): Fits to the UVS (Top), the research line fishing survey (Middle), and the mean age indices (Bottom) for the reefs in the Storm Cay region.

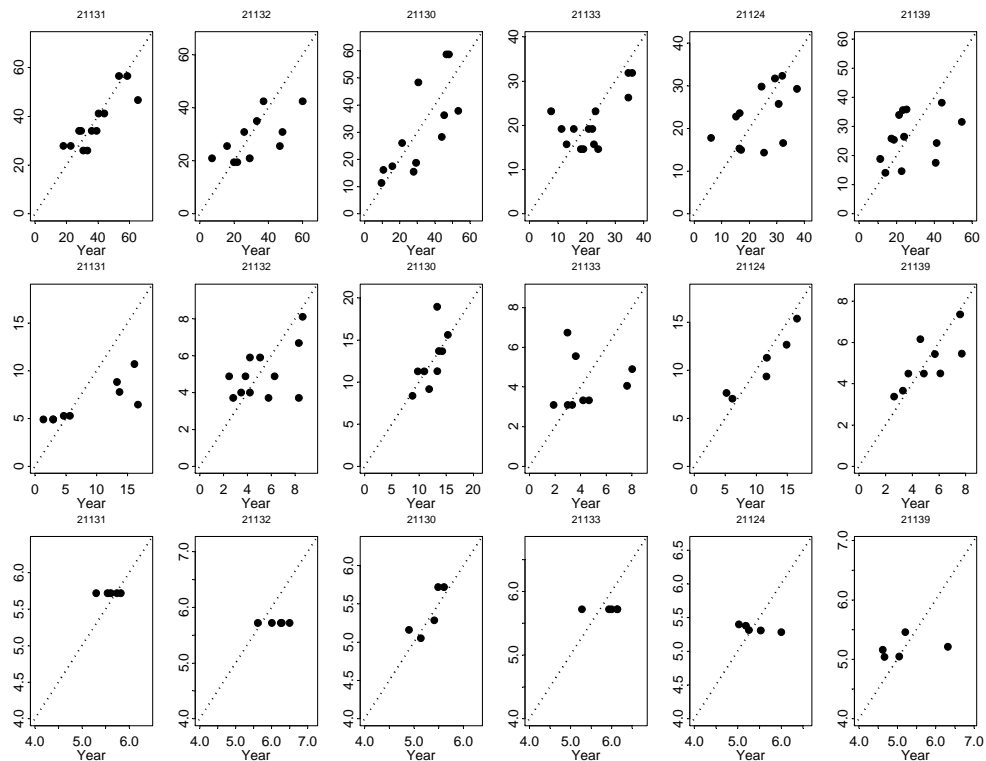


Table A3 summarizes the results from all eight analyses. The table lists, for the Marine National Park reefs opened to fishing ('MNP-Fished' reefs) in each region, the estimates and asymptotic coefficients of variation of: the exploitable biomass at the start of 1996 (B_{96}), the exploitable biomass at the start of 2000 (B_{00}), the ratio of B_{00} to B_{96} , expressed as a percentage, and the exploitation rate in April 1997. The ratio B_{00}/B_{96} represents slightly different things for the reefs pulsed in 1997 and 1999. For the 1997-pulsed reefs, the ratio represents the degree to which the biomass has rebuilt toward pre-harvest biomass (B_{96}) in the 2 years since closure following fishing. For the 1999-pulsed reefs, the ratio represents the degree to which the biomass has depleted relative to pre-harvest biomass (B_{96}). Table A3 also lists M , the objective function (see Equations A.3-A.5), the contribution to the value of the objective function by the UVS and research line fishing index data combined, and the contribution by the mean age data.

Exploitable biomass for the MNP-Control reefs is independent of time, which is also the case for the MNP-Fished reefs before they are 'pulse fished' in 1997 and 1999. This is because the estimator ignores stochastic fluctuations in population size (due, for example, to variation in recruitment and natural mortality) and the catches from closed reefs are assumed to be zero for the base-case analysis. The impact of "pulsing" of (opened) MNP reefs is most evident for Yankee reef and reef 21-130 (Fig. A3).

The fits to the data series are generally rather poor. This is primarily due to the lack of clear signals in the data. This is most evident for the Lizard region (Fig A4a, A5a), although the lack of data contrast for this region was anticipated soon after the attempt to pulse the MNP-Fished reef in this region in 1997-98 (Mapstone *et al.*, 1998). The impact of pulsing of MNP-Fished reefs in 1997-98 is evident in the UVS and research line fishing survey data for Yankee reef (Townsville region), reef 20-136 (Mackay region), and reef 21-130 (Storm Cay region) (Figures A4b – A4d; A5b – A5d). The model tends to mimic the trends in abundance indices for these reefs reasonably well. There is no evidence for the impact of pulse fishing on any of the GU-Fished reefs and the MNP reefs pulsed during 1999-2000. The fits to the data for reefs 20-137, 21-131 and 21-139 and Bax and Boulton reefs appear reasonable, although, of course, care needs to be taken when examining noisy fits visually.

The estimates of natural mortality for the base-case analysis range from 0.57yr^{-1} to 0.73yr^{-1} (Table A3a). These estimates are substantially larger than the 0.2yr^{-1} for age 4+ animals on which the simulations conducted by Punt *et al.* (2001) were based and the 0.3yr^{-1} used for ELFSIM. The estimates of M are determined primarily from the age-composition data for the unfished ("MNP-Control") reefs. However, the age-composition data for these reefs (Figure A6) drop off rapidly with age above the assumed age-at-recruitment of 4 years. Reasons other than high natural mortality for the age-profiles in Figure A6 include poaching on green reefs, declining selectivity with age, and the impact of delayed recovery of previously fished reefs (most of the MNP reefs were closed for many years prior to the ELF surveys commencing, but one was closed only about 2 years prior). If the estimates of M based on the unfished reefs in Table A3 are considered implausible, it may be necessary to pre-specify the value for M based on auxiliary information when applying estimators to the ELF experimental data and perhaps even ignoring the age-composition data altogether. It should be noted that Deriso-Schnute delay-difference models are all based on the assumption of flat selectivity above the age-at-recruitment. If estimators that allow selectivity to drop off with age are to be considered, this will necessitate the development of a completely new estimator.

The coefficients of variation for the estimates of biomass, depletion and exploitation rate (Table A3b) exhibit some general patterns. The coefficients of variation for the reefs that were fished in 1999 are higher than those for the reefs that were fished in 1997. This is not unexpected, because the recovery period for the reefs fished in 1997 is already underway, increasing the contrast in the data. The coefficients of variation for the reefs in the Lizard region are higher than those for the reefs in the other three regions. This is due to the particularly uninformative nature of the data for the Lizard region (Figures A4a and A5a). The coefficients of variation for the 'MNP-fished 1997' reefs in the Townsville and Storm Cay

regions are very small and should be viewed with circumspection – clearly these CVs underestimate the true extent of uncertainty. This is evident, for example, by comparing the point estimates of biomass, depletion and exploitation rate for these reefs across the sensitivity tests in Table A3b.

Figure A6: Age-frequency distributions (pooled over time and reefs) for the “MNP-Control” reefs in each region.

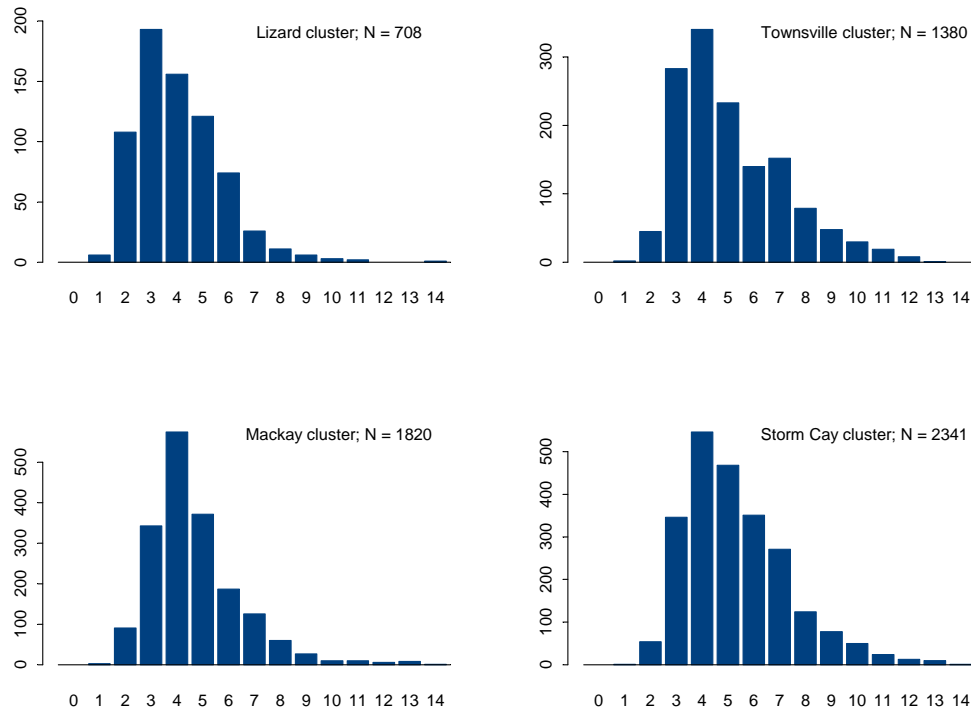


Table A2: Specifications of scenarios for the eight analyses of Experimental data: A **Base** case and 7 sensitivity analyses (2-8).

Specification	Alternative Run							
	Base	2	3	4	5	6	7	8
Catches on closed reefs	Ignored	Ignored	Included	Ignored	Ignored	Included	Ignored	Ignored
Schnute modification	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes
Mean age = absolute index	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes
Reefs assumed fished	GU only	GU only	GU only	All	None	All	MNP=0.8	MNP=0.6

Table A3: Results of the application of the estimator to the ELF experimental data given the specifications of the analyses provided in Table A2.

Table A3-a): Quantities common across regions. “**Index**”, “**Age**” and “**Total**” list respectively the contributions of the abundance index and age data to the objective function and the value of the objective function itself.

Run	Lizard Region				Townsville Region				Mackay Region				Storm Cay Region			
	<i>M</i>	Index	Age	Total	<i>M</i>	Index	Age	Total	<i>M</i>	Index	Age	Total	<i>M</i>	Index	Age	Total
Base	0.60	-91.8	82.4	-22.2	0.69	3.9	201.4	200.6	0.73	-7.5	56.4	51.1	0.57	36.6	83.5	123.0
2	0.16	-95.4	22.8	-86.6	0.47	-24.5	174.4	144.2	0.22	-13.4	51.2	38.0	0.25	29.3	52.3	80.3
3	0.58	-95.7	78.1	-27.7	0.63	1.4	172.8	177.1	0.73	-5.8	56.7	52.6	0.53	31.8	73.9	104.8
4	0.50	-92.5	50.0	-52.2	0.66	5.7	196.5	197.3	0.68	-9.6	51.7	44.4	0.51	39.1	68.8	110.0
5	0.65	-93.8	97.0	-9.7	0.70	7.0	204.3	212.6	0.81	4.5	105.6	109.0	0.58	36.9	87.8	125.9
6	0.46	-102.3	40.7	-68.4	0.59	3.6	161.9	169.2	0.68	-7.6	51.2	45.6	0.48	27.5	66.6	92.4
7	0.56	-91.8	86.0	-16.9	0.64	7.2	223.8	226.4	0.67	-9.5	57.7	50.7	0.52	42.2	90.0	135.1
8	0.50	-91.2	93.2	-6.8	0.6	11.9	253.5	261.0	0.60	-11.5	64.5	55.9	0.45	49.7	103.3	155.6

Table A3-b): Reef-specific biomass and exploitation rate estimates (and CVs). **Key:** B_{96} – Biomass in 1996, before pulse fishing; B_{00} – Biomass in 2000; B_{00}/B_{96} – Status of the stock after fishing, as % of B_{96} ; F_{97} – Fishing mortality on open reefs in 1997.

Run	Region	MNP-Fished 1997								MNP-Fished 1999					
		B_{96} (kg)		B_{00} (kg)		B_{00}/B_{96} (%)		F_{97}		B_{96} (kg)		B_{00} (kg)		B_{00}/B_{96} (%)	
Base	Lizard	6,119	0.57	5,986	0.57	97.8	0.58	0.006	0.01	6,722	0.66	5,663	0.78	84.3	0.12
	Townsville	5,522	0.06	4,051	0.06	73.4	0.08	0.305	0.03	6,046	0.23	3,833	0.36	63.4	0.13
	Mackay	7,423	0.31	6,955	0.31	93.7	0.33	0.188	0.02	31,924	0.43	29,372	0.47	92.0	0.04
	Storm Cay	8,280	0.10	6,053	0.08	73.1	0.11	0.144	0.03	10,189	0.48	10,189	0.48	100.0	0.00

Table A3-b) (Continued):

Run	Region	MNP-Fished 1997								MNP-Fished 1999					
		B_{96} (kg)		B_{00} (kg)		B_{00}/B_{96} (%)		F_{97}		B_{96} (kg)		B_{00} (kg)		B_{00}/B_{96} (%)	
2	Lizard	17,639	0.61	17,292	0.61	98.0	0.63	0.002	0.01	20,421	0.56	19,149	0.60	93.8	0.04
	Townsville	10,057	0.23	6,544	0.23	65.1	0.32	0.166	0.11	8,137	0.30	5,519	0.44	67.8	0.14
	Mackay	13,221	0.67	11,511	0.67	87.1	0.75	0.106	0.09	91,376	0.69	88,436	0.71	96.8	0.02
	Storm Cay	29,628	0.41	24,276	0.39	81.9	0.45	0.033	0.07	27,108	0.52	27,108	0.52	100.0	0.00
3	Lizard	13,245	0.47	14,200	0.49	107.2	0.45	0.003	0.03	29,855	0.41	27,589	0.44	92.4	0.03
	Townsville	6,011	0.05	3,820	0.04	63.5	0.08	0.321	0.04	4,994	0.12	2,231	0.28	44.7	0.16
	Mackay	8,439	0.30	7,572	0.30	89.7	0.33	0.169	0.03	33,008	0.45	30,665	0.49	92.9	0.03
	Storm Cay	12,499	0.13	7,921	0.08	63.4	0.13	0.12	0.05	10,933	0.46	10,933	0.46	100.0	0.00
4	Lizard	2,969	0.67	3,258	0.67	109.7	0.70	0.011	0.08	4,440	0.54	3,334	0.72	75.1	0.18
	Townsville	4,958	0.06	4,328	0.07	87.3	0.09	0.309	0.08	6,000	0.23	3,767	0.36	62.8	0.13
	Mackay	6,631	0.29	6,101	0.29	92.0	0.31	0.211	0.03	30,824	0.42	28,798	0.45	93.4	0.07
	Storm Cay	7,110	0.10	6,252	0.08	87.9	0.11	0.152	0.06	9,682	0.48	9,682	0.48	100.0	0.00
5	Lizard	6,338	0.53	6,217	0.53	98.1	0.54	0.006	0.01	6,686	0.63	5,646	0.75	84.5	0.12
	Townsville	5,617	0.06	4,192	0.06	74.6	0.09	0.3	0.03	6,018	0.21	3,817	0.33	63.4	0.12
	Mackay	5,981	0.27	5,590	0.27	93.5	0.29	0.234	0.02	15,353	0.55	12,845	0.66	83.7	0.11
	Storm Cay	8,308	0.11	6,124	0.08	73.7	0.11	0.143	0.03	9,096	0.48	9,096	0.48	100.0	0.00
6	Lizard	3,395	0.32	4,267	0.33	125.7	0.32	0.01	0.08	17,393	0.37	14,686	0.44	84.4	0.07
	Townsville	5,153	0.05	4,273	0.06	82.9	0.08	0.327	0.08	4,906	0.11	2,066	0.28	42.1	0.16
	Mackay	7,541	0.28	6,588	0.27	87.4	0.31	0.19	0.04	29,142	0.45	26,877	0.49	92.2	0.08
	Storm Cay	11,399	0.14	9,345	0.08	82.0	0.14	0.117	0.08	10,300	0.47	10,772	0.46	104.6	0.07
7	Lizard	4,267	0.58	4,647	0.58	108.9	0.60	0.008	0.02	4,376	0.62	3,825	0.80	87.4	0.18
	Townsville	5,076	0.05	4,048	0.05	79.7	0.08	0.313	0.03	5,229	0.22	3,601	0.36	68.9	0.14
	Mackay	6,104	0.28	6,265	0.28	102.6	0.30	0.216	0.03	27,760	0.42	28,425	0.46	102.4	0.04
	Storm Cay	7,432	0.09	5,823	0.07	78.4	0.10	0.154	0.03	8,509	0.48	9,572	0.48	112.5	0.00
8	Lizard	2,611	0.58	3,223	0.58	123.4	0.61	0.012	0.03	2,684	0.44	2,365	0.65	88.1	0.21
	Townsville	4,600	0.05	4,069	0.05	88.5	0.08	0.32	0.03	4,383	0.21	3,343	0.35	76.3	0.14
	Mackay	4,847	0.25	5,562	0.24	114.8	0.27	0.252	0.03	23,618	0.41	27,738	0.45	117.4	0.04
	Storm Cay	6,627	0.08	5,651	0.06	85.3	0.09	0.164	0.03	6,955	0.48	9,056	0.48	130.2	0.01

Figure A7 summarizes the residuals about the UVS / research line fishing indices and the mean age data by means of the histograms of residuals (standardised by their assumed standard deviations) and shows the histograms of the standard deviations of the standardised residuals. If the standard deviations assumed by the estimator (and by the protocol for testing estimators) were correct, the histograms of residuals should resemble a $N(0;1^2)$ distribution and the histograms of the standard deviations of the standardised residuals should be centred around 1. In contrast, the residuals are far more variable than expected from a $N(0;1^2)$ distribution and the standard deviations of the standardised residuals are typically 2 rather than 1. These results indicate that the assumed measurement error standard deviations are too small. Punt *et al.* (2001) show that a reduction in estimation ability is to be expected if the indices are less precise than expected (measurement error CVs of 0.2).

Figure A7 also shows the catchability coefficients for the UVS indices when these indices are fitted to the number of "legal" animals on each reef divided by reef perimeter. The values for the UVS catchability coefficients are generally lower than 1, which suggests that the assumption that reef perimeter provides a relatively good way to standardize abundances among reefs is perhaps not an ideal assumption.

The exploitation rates in April 1997 for the MNP-Fished reefs in the Townsville and Mackay regions (Yankee reef and reef 20-136) exceed 0.25 (Table A3b). These exploitation rates are not inconsistent with those that underlie the simulations conducted by Punt *et al.* (2001).

Figure A8 plots the UVS and research line fishing indices along with least squares fits (to aid the eye). The ideal relationship between the indices would be a straight line through the origin. The plots are complicated by error in the UVS indices and high observation error. This precludes definitive conclusions from these data. In contrast, the results of this figure further emphasize the importance of considering the form of the relationship between the indices and abundance explicitly. One consequence of the noisy relationships in Figure A8 and (in some cases) the lack of even a monotonic relationship between the UVS and research line fishing indices is that the estimates of γ (see Equation A.4) vary substantially among regions (-0.06, -0.03, 1.5, 0.04 for the Lizard, Townsville, Mackay and Storm Cay regions respectively). The γ values for the Lizard, Townsville, and Storm Cay regions imply that the model estimates that the research line fishing indices are essentially independent of abundances estimated by Underwater Visual Surveys (within the range of the data).

Figure A9 explores the relationship between the UVS and research line fishing indices further by fitting power relationships by means of standard linear regression and functional regression (assuming equal error variances on a log-scale). The relationships based on standard linear and functional regression differ substantially for some reefs, highlighting the importance of accounting for observation error in the UVS and research line fishing indices when attempting to identify the relationship between these indices. In general, the functional regression results suggest that research line survey indices change faster with abundance than the UVS indices.

Figure A7: Histograms of residuals (standardised by their assumed standard deviations) about the relative abundance indices, histograms of standardised residuals about the mean age indices, standard deviations of the standardised residuals about the relative abundance indices, standard deviations of the standardised residuals about the mean age indices, catchability coefficients for the UVS indices, and the annual year-specific residuals for Lizard (Top Left), Townsville (Top Right), Mackay (Bottom Left) and Storm Cay (Bottom Right).

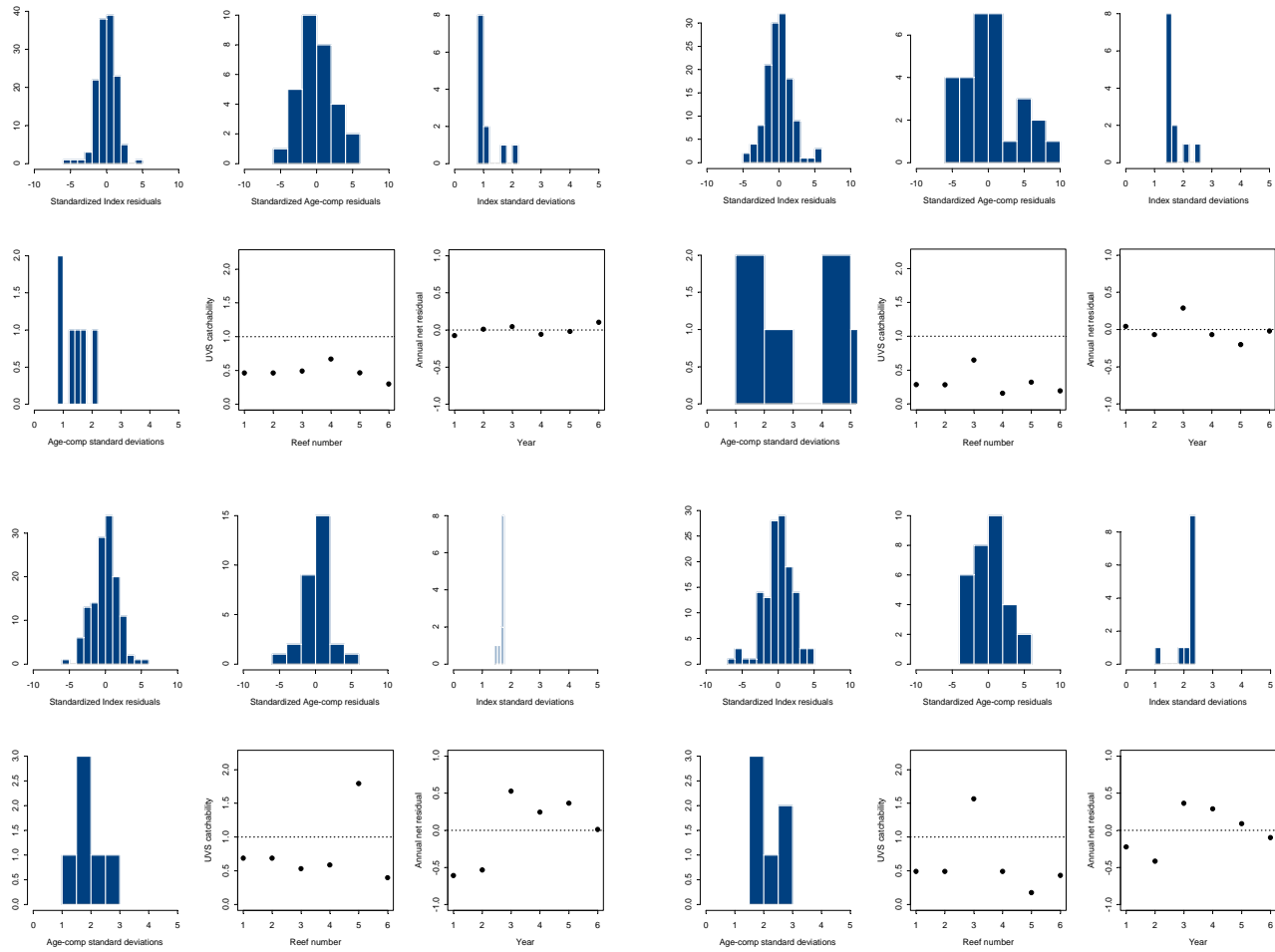


Figure A8: Correlations between the UVS (x-axis) and research line fishing catch survey (y-axis) indices of abundance over all surveys for each reef. The solid lines are least squares fits to the data.

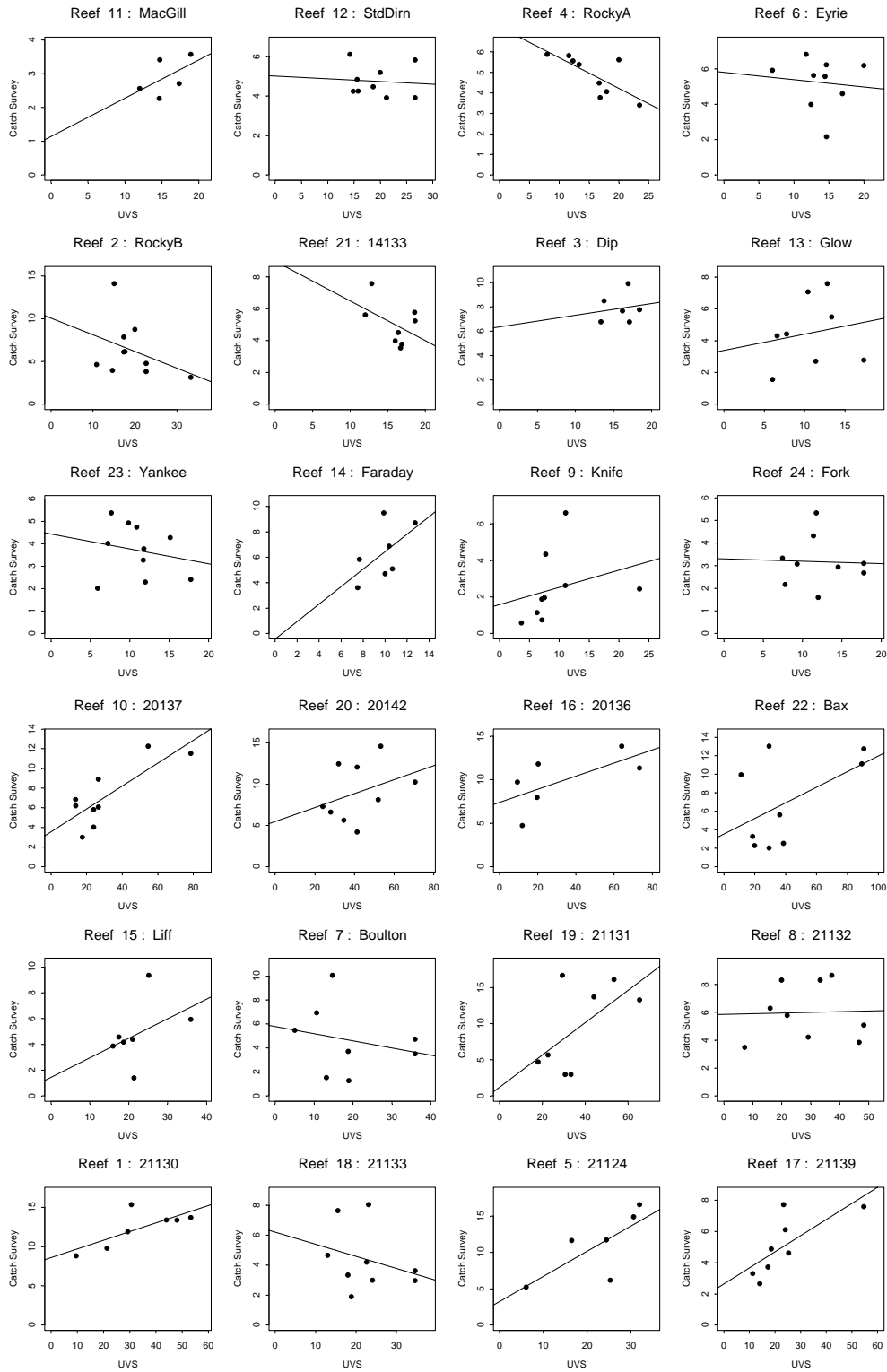
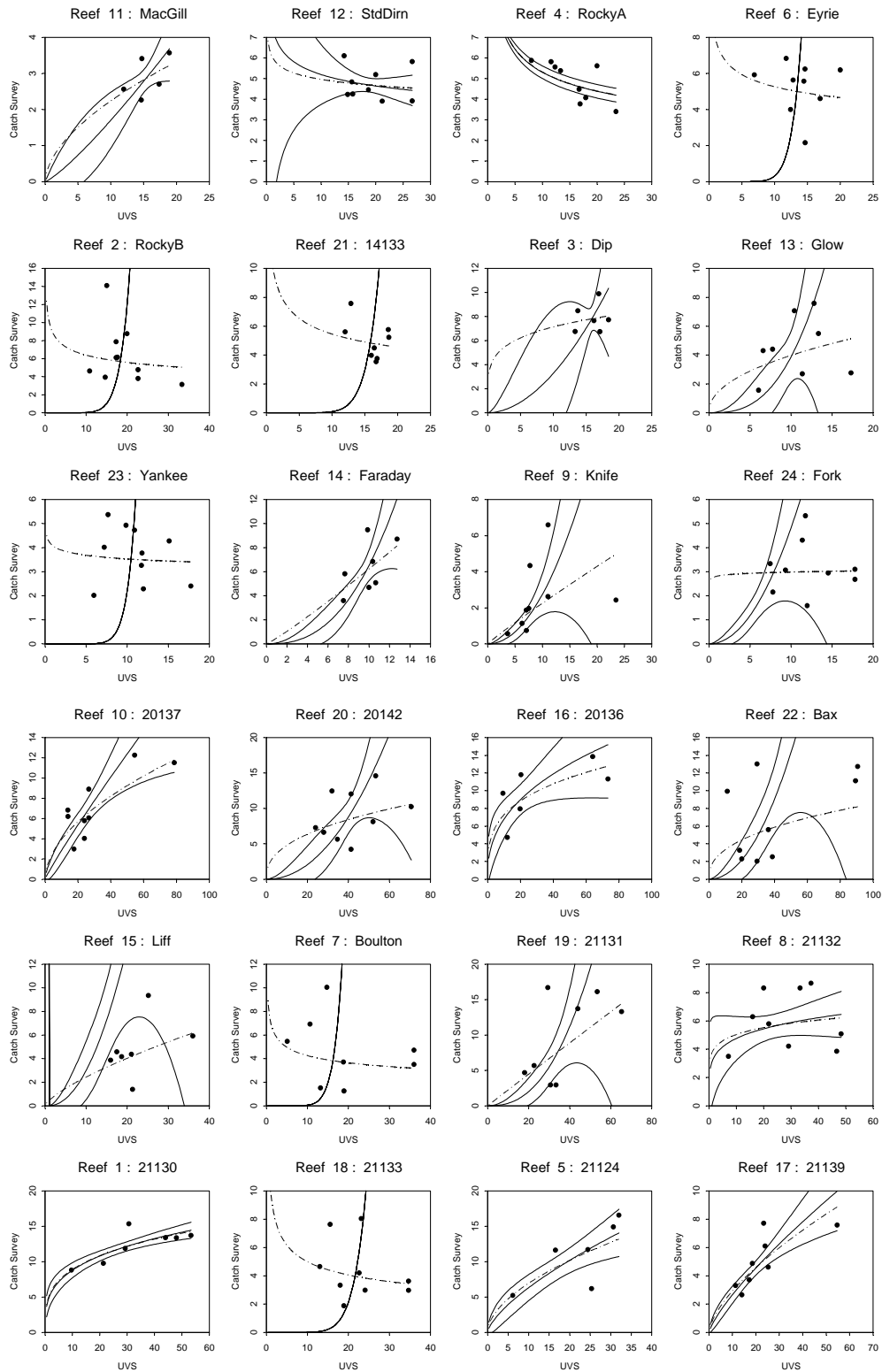


Figure A9: UVS and research line fishing catch survey indices for each reef. The solid and dotted lines are functional and standard linear regression fits to the data.



Sensitivity tests

Table A3 lists results for seven sensitivity tests in addition to those for the base-case analysis. The estimator selected by Punt *et al.* (2001) (run 2 in Table A3) leads to better fits to the data (indicated in Table A3a by lower values for the total objective function³). However, this is at the expense of larger mean catch weights (5-15kg per fish). Including the catches for the MNP Control reefs for all years in the analysis (run 3 in Table A3a) leads to slightly improved fits for the Lizard, Townsville, and Storm Cay regions.

The sensitivity to which reefs are assumed to be unfishable at the start of 1995 (and hence for which reefs an initial biomass needs to be estimated), is examined in runs 4, 5, 7, and 8. As expected, the lowest objective functions occur for run 4, which estimates a total of 6 initial biomasses for each region – the base-case analysis estimates only two initial biomasses for each region. The impact of this is most marked for the Lizard and Storm Cay regions and virtually non-existent for the Townsville region. Estimating a larger number of initial biomasses leads to lower values for M but, except for the Lizard region, the effect is relatively small compared to the difference between $0.2/0.3\text{yr}^{-1}$ and the estimates of M for the base-case analyses. This result suggests that the impact of fishing prior to the start of the experiment is not likely to be the reason for the lower average ages in the research survey catches. Fixing the initial (1995) biomasses for the MNP-Control and MNP-Fished reefs to 80% (run 7) and 60% (run 8) of the virgin level leads to larger values for the objective function. This is because, given the assumption of zero catches except when reefs are pulsed, recovery should be expected over the period 1995-2001 (compare the estimates of B_{00} / B_{96} for runs 7 and 8 with those for the base-case analysis in Table A3b). Recovery of biomass is, however, not consistent with the data available for these reefs.

The combination of sensitivity tests 3 and 4 is shown in run 6. The values for the objective function for this run are generally the lowest of those in Table A3a (ignoring the results for run 2), suggesting that there is probably value in including the reported catches for the MNP-Control and MNP-Fished reefs when computing biomasses, even though these catches may not all have been taken from closed reefs.

Recommendations for additional analyses

1. Information on selectivity from experimental fishing should be examined for the possibility that selectivity drops off with increasing age / size, resulting in over-estimates of M .
2. The performance of estimators that pre-specify rather than estimate M should be examined.
3. Simulations in which the amount of measurement error is defined so that the application of the current estimator leads to residuals with the amount of variation evident in, for example, Figure A7 should be conducted.
4. The simulations were parameterised using data on biological parameters (e.g. growth, selectivity) available in 1997. It may be appropriate to update the simulations based on the new biological information.
5. Future simulations (based on a revised simulation protocol) will allow the utility of estimators based on different data types (e.g. ignoring the mean age information) to be re-evaluated given the revised understanding of the uncertainties associated with the data.

³ No attempt will be made here to comment on the statistical differences among the runs because it is clear from Figures A4 and A7 that none of the models provide an adequate fit to the data, given their assumed residual standard deviations. Use of standard model selection criteria (e.g. AIC) in this case would inappropriately lead to rejection of simpler models at the expense of more complex models.

Annex A1: The Delay-Difference Estimator

The delay-difference models of Deriso (1980) and Schnute (1985) incorporate age-structure effects (subject to certain assumptions) and treat each component of production and mortality separately. Each parameter of this model has a specific biological interpretation. It is thus less subject to the criticism of simplicity often leveled at lumped biomass models, but nevertheless does not require a large amount of data for estimation purposes (only a time series of relative abundance indices, given independent availability of values for certain biological parameters). If desired, however, it can be tailored to make use of a wide variety of sources of data. For a monthly time-step, the population dynamics equation for the Deriso-Schnute model is given by:

$$B_{m+1} = B_m(1 + \rho)\lambda_m - \rho\lambda_m\lambda_{m-1}B_{m-1} + R_{m+1}w_r - \rho\lambda_{m-1}w_{r-1}/w_r R_m \quad (\text{A.1a})$$

$$N_{m+1} = N_m \lambda_m + R_{m+1} \quad (\text{A.1b})$$

$$\bar{a}_{m+1} = [(\bar{a}_m + 1/12)N_m \lambda_m + rR_{m+1}] / N_{m+1} \quad (\text{A.1c})$$

where:

- B_m is the (exploitable / recruited) biomass at the start of month m ,
- N_m is the number of (exploitable / recruited) animals at the start of month m ,
- \bar{a}_m is the mean age of recruited animals at the start of month m ,
- ρ is the Brody growth coefficient,
- λ_m is the survival rate during month m :

$$\lambda_m = e^{-M/12}(1 - C_m / B_m) \quad (\text{A.2})$$

- M is the instantaneous rate of natural mortality (assumed to be independent of age for ages above the age-at-recruitment),
- C_m is the catch during month m ,
- R_{m+1} is the number of recruits entering the fishery at the start of month $m+1$ (assumed to be independent of time, i.e. $R_m = R$),
- r is the age (taken to be 4yrs for this study) at which recruitment to the exploitable biomass occurs,
- w_r is the mass of a recruit (age r), and
- w_{r-1} is the mass of a recruit the week before it recruits.

The derivations for Equations (A.1a) and (A.1b) are given by Deriso (1980), Schnute (1985) and Walters (1986). Equation (A.1c) follows directly from Equation (A.1b), noting that animals are assumed to recruit to the fishery at age r .

The data included in the likelihood function are: (a) the indices of (adult) abundance from the visual and research line fishing surveys, and (b) the time series of the mean age of recruited (age $\geq r$) animals by month. The abundance indices based on the visual and research line fishing surveys are assumed to be log-normally distributed about the model estimates. The model also assumes that systematic variation in the values for the indices among reefs is a consequence of factors common to all reefs, which influence survey catchability, rather than "process error" effects such as systematic variation among reefs in natural mortality or larval settlement. The negative of the logarithm of the likelihood function (excluding constants independent of the model parameters) for the abundance index information is therefore given by:

$$L_1 = \sum_i \sum_v \sum_m \left\{ \lambda n \sigma_q + \frac{1}{2\sigma_q^2} \left(\lambda n I_m^{i,v} - \lambda n \hat{I}_m^{i,v} + \varepsilon_y \right)^2 \right\} \quad (\text{A.3})$$

where:

- $I_m^{i,v}$ is the observed index for index-type v (visual or research line fishing surveys) and reef i during month m ,
 - $\hat{I}_m^{i,v}$ is the model-estimate of the index for index-type v and reef i during month m :
- $$\hat{I}_m^{i,v} = q^{i,v} [(B_m + B_{m+1}) / 2]^{\gamma^v} \quad (\text{A.4})$$
- $q^{i,v}$ is the survey catchability coefficient for reef i and index-type v ,
 - ε_y accounts for sources of observation error which vary systematically among reefs (assumed to depend on year rather than on month),
 - γ^v is a factor to account for non-linearity in the relationship between research survey catch rate and abundance (γ is assumed equal to 1 for the index based on the visual surveys), and
 - σ_q is the standard deviation of the logarithms of the random fluctuations in survey catchability.

The information on the mean age of the catch is assumed to be a relative index of the model-estimate of mean age of the catch (see Equation A.1c). The negative of the logarithm of the likelihood function (excluding constants independent of the model parameters) for the mean age data is therefore given by:

$$L_2 = \sum_i \sum_m \left\{ \lambda n \bar{\sigma}_m^{i,obs} + \frac{(\bar{a}_m^{i,obs} - \tilde{q}^i \bar{a}_m^i)^2}{2(\bar{\sigma}_m^{i,obs})^2} \right\} \quad (\text{A.5})$$

where:

- $\bar{a}_m^{i,obs}$ is the observed mean age of recruited (age $\geq r$ years) animals on reef i during month m ,
- $\bar{\sigma}_m^{i,obs}$ is the standard deviation of $\bar{a}_m^{i,obs}$, defined using the equation:

$$\bar{\sigma}_m^{i,obs} = \sqrt{\frac{1}{n_m^i(n_m^i-1)} \sum_{a \geq r} n_{a,m}^i (a - \bar{a}_m^{i,obs})^2}$$

- n_m^i is the number of animals sampled from reef i during month m and aged to be age r or greater,
- $n_{a,m}^i$ is the number of animals sampled from reef i during month m and aged to be age a ($n_m^i = \sum_{a \geq r} n_{a,m}^i$), and
- \tilde{q}^i is the constant of proportionality between the observed and model-predicted mean ages.

Term L_2 assumes that the mean age information is a relative index of the mean age of the population (i.e. $\tilde{q}^i \neq 1$). Equation (A.5) is based on the assumption that mean age of the catch is normally distributed. This assumption differs from the conventional assumption made when conducting statistical catch-at-age analysis that the proportion of the catch in a given age-class is multinomially distributed (e.g. Fournier and Archibald 1982) because the delay difference model does not keep explicit track of the number of animals in each age-class.

The negative of the logarithm of the likelihood function also includes a constraint on the year-specific deviations in catchability and a constraint on the values for the catchability coefficients for the visual surveys:

$$L_3 = \sum_y \left\{ \lambda n \sigma_r + \frac{\varepsilon_y^2}{2\sigma_r^2} \right\} \quad (\text{A.6})$$

$$L_4 = \sum_i \left\{ \lambda n \tilde{\sigma}_q + \frac{(\lambda n q_i^{obs} - \lambda n (z q^{i,1}))^2}{2\tilde{\sigma}_q^2} \right\} \quad (\text{A.7})$$

where:

- z is the mean of the catchability coefficients for the visual surveys,
- σ_r is the standard deviation of the ε_y 's,
- q_i^{obs} is a relative measure of the catchability coefficient for the visual survey index for reef i (the perimeter of the reef), and
- $\tilde{\sigma}_q$ is the standard error of the logarithm of q_i^{obs} .

It is assumed that the value of ρ is known and equal to 0.99999 and those of σ_q and $\tilde{\sigma}_q$ are known and equal to 0.2 and 0.4 respectively. The value of M is treated as an estimable parameter. For the reefs that are 'closed' at the start of the experiment, the population is assumed to be in equilibrium while for the reefs that are 'open' at the start of the experiment, the equilibrium biomass is multiplied by a reef-specific factor B_1 . The model is fitted to the data for each of the regions separately. The free parameters of the model are listed in Table A1-1.

Table A1-1: The free parameters of the model. The number of parameters refers to a (base-case) fit to the information for a region of six reefs.

SYMBOL	Description	N° parameters
R	Recruitment	6
B_1	Ratio of initial to virgin biomass	2
σ_r	Extent of systematic variation in catchability*	1
γ	Non-linearity factor	1
q	Survey catchability coefficients	12
z	Mean survey catchability (visual surveys)	1
ε_y	Observation error deviations*	each year
\tilde{q}	Constant of proportionality – age data*	6

* The estimates for these parameters can be obtained analytically as a function of those for the remaining parameters.

Appendix B: The ELFSim Operating Model

The Biological Components of the Operating Model

The biological component of the operating model incorporates many of the features of the models of coral trout population dynamics developed by Walters and Sainsbury (1990) and Mapstone *et al.* (1996e). The coral trout resource on the Great Barrier Reef is assumed to consist of several post-larval, post settlement sub-populations. Each sub-population is associated with a single reef. The population dynamics model is age-, sex- and size-structured, assumes that the number of 0-year-olds is related to the size of the reproductive component of the population according to a form of the Beverton-Holt stock-recruitment relationship, and allows for larval movement among post-settlement sub-populations. Several sources of process error (Francis and Shotton 1997) such as variation in natural mortality and larval survival are included in the operating model. The operating model allows for multiple effort-classes. The current implementation includes three effort-classes (commercial effort: effort-class 0, charter effort: effort-class 1, and recreational effort: effort-class 2).

The size-structure of the population is modelled by dividing each cohort into ten growth groups. All animals within a growth group are assumed to grow according to the same growth curve, but growth curves differ among growth groups. The model allows for movement of larvae but ignores the possibility of movement of animals aged 1 and older, there being little evidence for movement of such animals (Davies 1995a,b). In addition, ignoring such movement is conservative because movement of 1+ animals among reefs would decrease recovery times for depleted populations.

The equations below assume that the parameters determining natural mortality, fecundity, sex-change and growth are independent of reef. The software that implements the model has the functionality to allow these parameters to depend on reef, but reef-dependence has been ignored in this Appendix for ease of presentation.

Basic Population Dynamics

The resource dynamics are modelled using the equations:

$$N_{y+1,a}^{r,k} = \begin{cases} N_{y+1,a}^{r,k} & a = 0,1 \\ N_{y,12,a-1}^{r,k} e^{-Z_{y,12,a-1}^{r,k}} & a = 2, \dots, x-1 \\ N_{y,12,x-1}^{r,k} e^{-Z_{y,12,x-1}^{r,k}} + N_{y,12,x}^{r,k} e^{-Z_{y,12,x}^{r,k}} & a = x \end{cases} \quad (\text{B.1})$$

where:

$N_{y,a}^{r,k}$ is the number of fish of age a in growth group k on reef r at the start of year y ,

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

$$N_{y,m+1,a}^{i,j,k} = N_{y,m,a}^{i,j,k} e^{-Z_{y,m,a}^{i,j,k}} \quad (\text{B.2})$$

$Z_{y,m,a}^{r,k}$ is the total mortality on fish of age a in growth group k on reef r during month m of year y .

$$Z_{y,m,a}^{r,k} = M_{y,a} / 12 + \sum_v F_{y,m,a,v}^{r,k} \quad (\text{B.3})$$

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

$F_{y,m,a,v}^{r,k}$ is the fishing mortality on fish of age a in growth group k on reef r during month m of year y by vessel-class v , and

x is the maximum age considered (taken to be a “plus group”).

The maximum age x (18 yr – Table B1) has little impact on the results because the rate of natural mortality assumed for animals aged 2 and older (0.3 yr^{-1}) implies that very few coral trout reach age 18.

0-Year-Olds

All fish are born as females. The number of 0-year-olds on reef r at the start of year y^4 is determined from a contribution from spawning on reef r and from a contribution from all other reefs (Mapstone *et al.* 1996):

$$N_{y,0}^{r,k} = K^k (ss \mathcal{F} S_y^r + c^r BL_y^r) \quad (\text{B.4})$$

where:

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

$$S_y^r = \sum_{a=1}^x \sum_k f_{L_{k,a}} w_{L_{k,a}} N_{y,a}^{r,k} (1 - P_{L_{k,a}}) \quad (\text{B.5})$$

\mathcal{F} is the maximum rate of larval survival to settlement in the absence of density-dependent mortality of larvae, multiplied by the average number of eggs per kg body weight for reef r ,

$L_{k,a}$ is the length of a fish of age a in growth group k ,

ss is the proportion of larvae that settle on the same reef as they were spawned,

K^k is the fraction of larvae that fall into growth group k ,

w_L is the mass of a fish of length L ,

f_L is the fraction of animals of length L that are mature,

P_L is the probability that a fish of length L is male,

BL_y^r is the background supply of larvae to reef r from all reefs during year y :

$$BL_y^r = \sum_{r'} \mathcal{F} S_y^{r'} \Omega^{r',r} \quad (\text{B.6})$$

c^r is the scaling factor for reef r to account for variation in background larval supply among reefs, and

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

The values in the larval dispersal matrix, Ω , are proportional to the fraction of larvae that move from reef r' to reef r because the value for c^r provides an overall scaling factor. The values in the larval dispersal matrix are determined using one of three approaches:

- “Uniform” distribution of larvae: *i.e.*, $\Omega^{r',r} = 1$.
- Pre-specified: *i.e.*, The values for the $\Omega^{r',r}$ are determined directly from models of larval movement (James *et al.* 2002).
- Distance-based distribution of larvae: *i.e.*, $\Omega^{r',r} = \exp(-3.4 \text{dist}(r, r') - 3.91)$ where the function *dist* determines the distance between the centroids of reefs r and r' . This relationship is based on fitting a linear model to the logarithm of fraction of the larvae which move from one reef to another from the model of larval movement.

The bulk of the analyses are based on the distance-based approach (approach c) because a uniform distribution of larvae (b) is unrealistic, particularly when the model is applied to a large geographic area. Use of larval dispersal rates determined from models of larval advection and larval behaviour are clearly desirable, but at present such a model exists for only a small fraction of the Great Barrier Reef (321 reefs in the Cairns Section) and does not

⁴ Births are assumed to occur at the start of the year following Ferriera and Russ (1994) and Russ *et al.* (1996).

include all the reefs in the ELFSim model (e.g., no virtual reefs are included in the dispersal models of James *et al.* 2002). The dedicated fine-scale hydrodynamic and larval advection models are being extended to encompass the entire GBR and will facilitate future use of approach b) in ELFSim.

The value for $f\%$ is determined from the parameters ss , st and the age-structure of the population on reef r at (deterministic) pre-exploitation equilibrium:

$$f\% = \frac{st N_{0,0}^r}{ss S_0^r} \quad (B.7)$$

where:

- S_0^r is the size of the reproductive component of the population on reef r at pre-exploitation equilibrium,
- st is the fraction of the larvae that settle on reef r that originated from reef r ,
- $N_{0,0}^r$ is the number of 0-year-olds on reef r at pre-exploitation equilibrium.

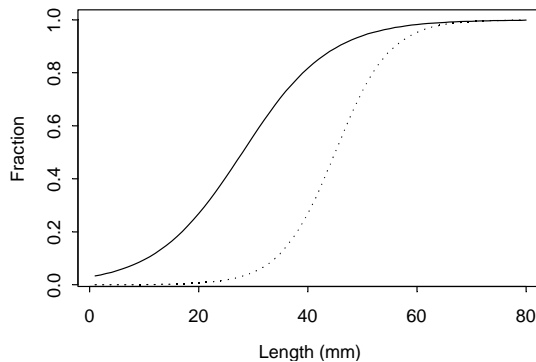
The values for the reef scaling parameters, c^r , are then determined using the equation:

$$c^r = \frac{ss (1-st) f\% N_{0,1}^r}{st \sum_{r'} f\% N_{0,1}^{r'} \Omega^{r',r}} \quad (B.8)$$

where $N_{0,1}^r$ is the number of 1-year-olds on reef r at pre-exploitation equilibrium.

The value of c^r (see Annex B1 for derivation) depends on the larval dispersal matrix. The value of c^r is recalculated annually for scenarios in which the larval dispersal matrix is based on the model of larval advection and behaviour, and hence varies among years.

Figure B1: Fraction of animals that are mature (solid line) and male (dotted line) by length.



The fraction of the animals that are mature, f_L , and the fraction of animals that are male, P_L are determined from logistic functions of length (Fig. B1; Table B2).

Recruitment to Reefs

The number of 1-year-olds in growth group k on reef r at the start of year $y+1$ is the number of zero-year-olds in growth group k on reef r the previous year modified by the density-dependent mortality between ages 0 and 1 plus the impact of random environmental variability and 'recruitment pulses'⁵:

$$N_{y+1,1}^{r,k} = N_{y,0}^{r,k} e^{M_{y,0} - \beta^r (U_{y+1}^r / U_0^r - 1)} e^{\varepsilon_y^r - \sigma_r^2 / 2} e^{\sum_i x_{y,i} \exp(\omega_p \text{dist}(r, c_i))} \quad (\text{B.9a})$$

$$U_{y+1}^r = \sum_k (N_{y,0}^{r,k} e^{-M_{y,0}} + \sum_{a=2}^J N_{y+1,a}^{r,k}) \quad (\text{B.9b})$$

$$\varepsilon_y^r = \tau_r z_y + \sqrt{1 - \tau_r^2} z_y^r \quad (\text{B.9c})$$

where:

- β^r is the density-dependence parameter for reef r ,
- U_0^r is the value of U_y^r at pre-exploitation equilibrium,
- J is the maximum age of a 'juvenile',
- z_y, z_y^r are iid random deviates from $N(0; \sigma_r^2)$,
- σ_r^2 is the overall inter-annual variation in larval abundance,
- τ_r is the correlation in larval abundance among reefs,
- $x_{y,i}$ is the magnitude of the i 'th 'recruitment pulse' during year y , generated from the normal distribution, $N(0; 1^2)$,
- ω_p is the parameter that determines the spatial extent of a 'recruitment pulse', and
- c_i is the center of the i 'th 'recruitment pulse'.

Given the formalism adopted, 'recruitment pulses' can lead to higher or lower than expected survival rates from age 0 to age 1. The centres for the 'recruitment' pulses are distributed randomly over the Great Barrier Reef. Note that if the model is run for a subset of the GBR, it is possible that the centres for some of the 'recruitment pulses' may fall outside the area considered in the model.

The value for the parameter β^r is determined by solving a system of equations for a pre-specified value for the steepness of the stock-recruitment recruitment, h (Annex B2). Steepness is defined after Francis (1992) to be the fraction of the (average) pre-exploitation number of 1-year-olds to be expected when the spawner biomass is reduced to 20% of its (average) pre-exploitation level. Four levels for steepness (0.35, 0.5, 0.65 and 0.8) were considered (Appendix C), with a value of 0.5 being used in most simulations.

⁵ The noise terms also can be considered to impact larval mortality rather than that between ages 0 and 1.

Natural mortality

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

$$M_{y,a} = (M_{y-1,a})^{\tau^M} (M'_{y,a} e^{\varepsilon_{y,a}^M - (\sigma^M)^2/2})^{\sqrt{1-(\tau^M)^2}} \quad \varepsilon_{y,a}^M \sim N(0; (\sigma^M)^2) \quad (\text{B.10a})$$

$$M'_{y,a} = \begin{cases} M_a + M_c \eta_y & \text{if } y \leq y_{fst} \\ M_a + M_{fin,a} \frac{(y-y_{fst})}{(y_{lst}-y_{fst})} + M_c \eta_y & \text{if } y_{fst} < y < y_{lst} \\ M_a + M_{fin,a} + M_c \eta_y & \text{otherwise} \end{cases} \quad (\text{B.10b})$$

where:

- M_a is the expected rate of natural mortality on animals of age a ,
- σ^M is the parameter that determines the extent of temporal variation in natural mortality,
- τ^M determines the extent of temporal correlation in natural mortality,
- y_{fst} is the year in which the natural mortality rate begins to change,
- y_{lst} is the year after which the natural mortality rate ceases to change and remains constant,
- $M_{fin,a}$ is the amount by which natural mortality changes for an animal of age a ,
- M_c is the amount by which natural mortality increases during a catastrophic event,
- η_y is a random variable that is 1 with probability p_c and 0 otherwise, and
- p_c is the probability of a catastrophic event.

Equation (B.10) allows for catastrophic events (such as the impact of a cyclone) to increase natural mortality on all fish by $M_c \text{ yr}^{-1}$. The probability of a catastrophic event is assumed to be p_c (base-case value zero). The value of η_y is independent of reef so that it is assumed that a catastrophic event has the same impact across all of the reefs included in the model. Time-trends in natural mortality cause natural mortality for age a to increase from M_a to $M_a + M_{fin,a}$ over the years y_{fst} to y_{lst} . This formulation provides a framework within which some of the possible impacts of global climate change on the dynamics of and fishery for coral trout can be investigated if credible parameterisations were available. The base-case trial ignores the possibility of climate change. The base-case values for the remaining parameters that determine natural mortality (M_a , σ^M and τ^M) are listed in Table B1. The values for natural mortality-at-age are guestimates based roughly on the assumption that natural mortality should decline with age, and the observed age-/size-structure of coral trout on closed reefs.

Table B1: The base-values for the fixed parameters of the operating model. The values for the parameters related to changes over time in natural mortality and to catastrophic events are not listed as these factors are not part of the base-case analyses.

Parameter	Base-case value	Source
Maximum age– x	18 yr	Mapstone <i>et al.</i> 1996, Ferreira and Russ 1994.
Mortality @ Age 0	0.4 yr ⁻¹	Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001.
Mortality @ Age 1	0.35 yr ⁻¹	B. Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001
Mortality @ Age 2+	0.3 yr ⁻¹	B. Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001
Temporal variation in natural mortality - σ^M	0.05	Mapstone <i>et al.</i> 1996
Temporal auto-correlation in natural mortality - τ^M	0	Assumed
Variation in 0-year-old survival - σ_r	0.6	Mapstone <i>et al.</i> 1996, Campbell <i>et al.</i> 2001.
Spatial correlation in 0-year-old survival - τ_r	0.5	Assumed
Larval self seeding – st	0.1	Jones <i>et al.</i> 1999, James <i>et al.</i> 2002.
Larval retention probability - ss	0.05	Mapstone <i>et al.</i> 1996
Maximum age of a ‘juvenile’ – J	18	Mapstone <i>et al.</i> 1996
Steepness, h	0.5	Pre-specified*
Length-mass parameters, $\lambda_1(b_1), b_2$	-11.03, 2.97	G.R. Russ pers. com, Mapstone unpub. data.
Extent of density-dependence in catchability - ϕ	0	Assumed
Variability in effort-fishing mortality relationship - σ_ζ	0.3	Assumed

* Results of these sensitivity analyses for alternative values of these parameters are summarised in Appendix C.

Table B2: Parameters related to selectivity, maturity, and the proportion male / sex-change.

Parameter	Base-case value	Source
Length-at-50%-selectivity (FL)	322 mm TL	Fulton 1996, Fulton <i>et al.</i> 1999
Length-at-95%-selectivity (FL)	375 mm TL	Fulton 1996, Fulton <i>et al.</i> 1999
Length-at-50%-maturity (FL)	280 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.
Length-at-95%-maturity (FL)	360 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.
Length-at-50%-sex change	450 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.
Length-at-95%-sex change	500 mm FL	Adams 1996, 2002, Adams <i>et al.</i> 2001.

Growth

The growth of an individual is assumed to be governed by the von Bertalanffy growth equation:

$$L_{k,a} = L_{\infty}^k (1 - e^{-\kappa^k (a-t_0^k)}) \quad (\text{B.11})$$

Variation in growth among individuals is modelled by assuming that the parameters κ , L_{∞} and t_0 differ among growth groups but that all animals within a growth group grow according to the same growth curve. The values for the parameters that determine growth (*i.e.* the values for κ , L_{∞} and t_0) and those that determine the proportion of 0-year-olds in each growth group (*i.e.* the values for the K^k) are determined by fitting a model to data collected on length-at-age, after accounting for gear selectivity (Annex B3).

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

$$w_L = b_1(L)^{b_2} \quad (\text{B.12})$$

where b_1, b_2 are the parameters of the relationship between length and mass (Table B1).

The Harvest Components of the Operating Model

The harvest components of the ELFSim operating model comprise two main processes: allocating effort over the simulation domain, and ‘harvesting’ catch from the reef-specific post-settlement populations.

Effort Allocation

The effort allocation algorithm in ELFSim determines the spatial distribution of fishing effort at each time step, given a specification for the total fishing effort by time step. This algorithm is not intended to mimic the individual decisions of skippers but instead represents the net effect of these actions at the fleet (effort-class) level. By applying the effort allocation algorithm to the entire effort-class or to sub-groups of effort-classes in particular regions, both data requirements and computer search time are reduced without necessarily sacrificing the ability of the model to mimic historical and predict future distributions of effort.

Briefly, the algorithm works for a given region and effort-class by ranking the sites (6' x 6' map references by which catch and effort is reported in commercial and charter fishing logbooks) in that region according to their historical catch rates, and then distributing the total effort for the region starting with the highest-ranked site. This algorithm therefore, depends on the assumption that fishers seek to maximise their catch rates. Limits are placed on the amount of effort that can be allocated to each site.

The effort allocation algorithm can be applied to different effort-classes (commercial, charter and recreational), and to different regions. The regions need not be separate. For example, to model the behaviour of vessels that fish the entire GBR (the ‘transient’ vessels), the region is taken to be the entire GBR.

The information supplied to ELFSim is the level of effort (by effort-class and region) for the first year of the projection period (1999) and either the rate at which annual effort increases, or the specified proportion of the initial (currently 1999) effort to be allocated after some time period. For example, effort may be set to 150% of the initial effort after five years. This last option is used to avoid discontinuities in effort between the historical and projection periods, by allowing a gradual change in total effort over time. Effort is assumed to change linearly with time.

The proportion of total effort allocated to ‘virtual’ reefs is constrained to the historical proportion from 1989 to 1998, because of a lack of empirical information on the amount of habitat on such reefs.

The total annual effort for each year of the projection period is divided among to the time-steps within the year to capture seasonal patterns. Seasonality in fleet effort is based on the distribution of effort by time step for the years for which real data are available. The allocation process for a future year (t) involves selecting a year at random from the appropriate period for each effort class (1989–98 commercial, 1996–1998 charter, and 1998 recreational), calculating the fraction of effort by time step for that year, and using these fractions to distribute total effort for projection year t to each time step.

ELFSim allocates effort to sites based on a four-step procedure:

- a) The sites that were fished in the previous time step are ranked according to past CPUE.
- b) Beginning with the top-ranked site, effort (based on the weighted average historical effort – see below for the details of how historical effort is weighted) is allocated until either there is no effort left, or all of these sites have been assigned some effort
- c) If effort remains to be allocated after step b), it is allocated to the sites that were not fished in the previous time step, but have been fished at some prior time prior. As in step b), these remaining sites are ranked according to historical CPUE, and effort is allocated based on the weighted average historical effort in the site, until either there is no effort left, or all of these sites have been assigned some effort
- d) If effort remains to be allocated after steps b) and c), it is allocated randomly to sites in small (5%) proportions. This step allows for exploration of new fishing grounds and is the only way that effort can be allocated to sites that have not been fished historically.

This algorithm depends on the historical CPUE for a site. The historical CPUE for a site in time step t used to allocate effort is calculated initially from the data for all previous years. The CPUE is then updated at the end of each time-step. This updating process down-weights older data using the formula:

$$\overline{CPUE}_t = \overline{C}_t / \overline{E}_t \quad (\text{B.13})$$

where:

$$\overline{CPUE}_t \quad \text{is the historical CPUE after time step } t^6, \quad \overline{C}_t = \delta \overline{C}_{t-\Delta t} + C_t \quad (\text{B.13a})$$

$\overline{C}_{t-\Delta t}$ is the discounted total catch one year before time step t ,
 C_t is the catch during time step t ,

$$\overline{E}_t = \delta \overline{E}_{t-\Delta t} + E_t \quad (\text{B.13b})$$

$\overline{E}_{t-\Delta t}$ is the discounted effort one year before time step t ,
 E_t is the effort during time step t , and
 δ is a discount factor.

The amount of effort assigned to a site depends on its average historical effort, and its “management status”. The weighted average historical effort for a given site and month is the product of the average historical effort, E_t^H and concentration factor, J . E_t^H is based on the site’s cumulative effort, $E_t^T = \sum_{t' < t} E_{t'}$, and the number years that the site has been fished in each month, N_t^F . The value for E_t^H depends on the whether the site was fished in the “previous month” (Equation B.14a) or not (Equation B.14b).

⁶ The dependence of these variables on site has been suppressed here for ease of presentation.

$$E_t^H = \begin{cases} 0.5 \left[E_t + \frac{E_t^T - E_t}{N_t^F - 1} \right] & \text{(A.14a)} \\ \frac{E_t^T}{N_t^F} & \text{(A.14b)} \end{cases}$$

ELFSim can be run defining the “previous month” for month t as one month before month t or the same month as month t the previous year. For the second of these options, the cumulative effort is computed for the same months as month t . The latter option was used for the simulations in this report.

ELFSim has the ability to concentrate effort in the sites with the highest CPUE. The concentration factor for site g is given by:

$$J_g = \left(C + \frac{1}{r_g} \right)^\alpha \quad \text{(B.15)}$$

where:

- r_g is the rank of site g ,
- α is the effort concentration parameter, and
- C is a constant such that:

$$\sum_g \left(C + \frac{1}{r_g} \right)^\alpha = G \quad \text{(B15b)}$$

where G is the number of sites.

A value of $\alpha=1$ implies that for $C=0.5$, the top-ranked site is given 150% of its average effort, whereas the 10th ranked site is given only 60% of its average effort. The base-case simulations ignore concentration of effort (*i.e.* $\alpha=0$).

Site management status (*e.g.* open or closed to fishing) is a variable in ELFSim specified by a “management status number”, M , between 0 (open) and 1 (closed). Values between 0 and 1 represent closed sites (or reefs) with some level of infringement. The effort assigned to a site is calculated as the average historical effort multiplied by $1-M$.

ELFSim allows for both spatial (edge effects) and temporal components to infringement of closed areas. The edge effort for a closed site g is calculated as:

$$I_g = \frac{1}{4} \sum_i I_{g,i} = \frac{1}{4} \sum_i \left[1 + 99e^{-rd_i} \right]^{-1} \quad \text{(B.16)}$$

where:

- d_i is the distance from site g to the edge of the closed area in direction i (north, south, east and west), and
- r is the parameter that specifies the rate with which infringements change with distance (base-case value 2.3). The base-case value implies a 50% reduction of infringement 3 sites (18 miles) from the edge of a closed area.

The temporal component to infringement of a closed site g is calculated as:

$$H_g = 1 - \left[1 + 99e^{-st} \right]^{-1} \quad \text{(B.17)}$$

where:

- t is the time (years) that site g has been closed, and
- s is the parameter that specifies the rate at which infringement changes with time (base-case value 2.3). The base-case value implies a 50% reduction of infringement 3 years since the area was closed.

An open site g with average effort x would, therefore, be allocated $J_g x$ days of effort, a closed site with full enforcement would be allocated 0 days, and a closed site with a management status of 0.5 would be allocated $J_g I_g H_g x$ days of effort. By varying M with time, rotational or seasonal closures of sites can be implemented on an annual basis as well as monthly.

Catches

The catch (in mass) of fish from reef r during month m of year y by effort-class v , $C_{y,m,v}^r$, is computed using the equation:

$$C_{y,m,v}^r = \sum_{a=0}^x \sum_k \frac{W_{L_{k,a+(m-0.5)/12}} D_{L_{k,a+(m-0.5)/12}} F_{y,m,a,v}^{r,k}}{Z_{y,m,a}^{r,k}} N_{y,m,a}^{r,k} (1 - e^{-Z_{y,m,a}^{r,k}}) \quad (\text{B.18a})$$

$$F_{y,m,a,v}^{r,k} = V_{L_{k,a+(m-0.5)/12}} F_{y,m,v}^r \quad (\text{B.18b})$$

where:

- V_L is the selectivity of the gear on fish of length L ,
- D_L is the fraction of animals of length L that are retained following capture,
- $F_{y,m,v}^r$ is the “fully-selected” fishing mortality applied to reef r by effort-class v during month m of year y :

$$F_{y,m,v}^r = q_v^r (B_{y,m}^r / B_0^r)^\phi E_{y,m,v}^r e^{\zeta_{y,m,v}^r - \sigma_\zeta^2 / 2} \quad (\text{B.19})$$

- $B_{y,m}^r$ is the biomass on reef r at the start of month m of year y available to the fishery (the exploitable biomass):

$$B_{y,m}^r = \sum_{a=0}^x \sum_k W_{L_{k,a+(m-0.5)/12}} D_{L_{k,a+(m-0.5)/12}} V_{L_{k,a+(m-0.5)/12}} N_{y,m,a}^{r,k} \quad (\text{B.20})$$

- B_0^r is the value of $B_{y,m}^r$ at the pre-exploitation equilibrium level,
- ϕ is a parameter that permits catchability to be density-dependent,
- $E_{y,m,v}^r$ is the effort applied by effort-class v on reef r during month m of year y :

$$E_{y,m,v}^r = \sum_b \frac{B_{y,m}^{r,b} E_{y,m,v}^b}{\sum_{r'} B_{y,m}^{r',b}} \quad (\text{B.21})$$

- $E_{y,m,v}^b$ is the effort applied by effort-class v in block b during month m of year y (determined by the effort dynamics model),
- $B_{y,m}^{r,b}$ is the biomass in block b during month m of year y available to the fishery that originates from reef r .

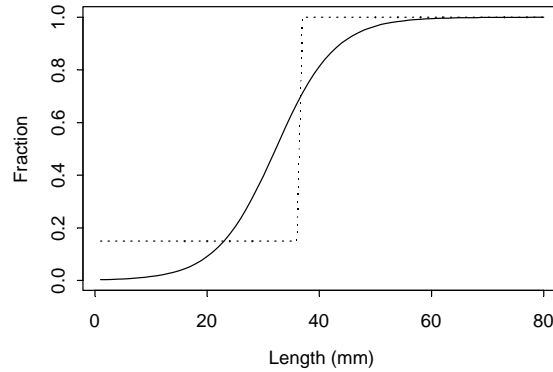
$$B_{y,m,v}^{r,b} = P^{r,b} B_{y,m,v}^r \quad (\text{B.22})$$

- $P^{r,b}$ is the proportion of reef r that is in block b .
- $\zeta_{y,m,v}^r$ is a factor to account for random variation in catchability ($\zeta_{y,m,v}^r \sim N(0; \sigma_\zeta^2)$), and
- q_v^r is the catchability coefficient for effort-class v and reef r .

In an ideal world, all under sized animals should be released alive following capture so that V_L should be a knife-edged function at the minimum size (36cm Fork Length, 38cm Total Length). Some under sized animals will die following release, however, while some under sized coral trout will be marketed illegally. Selectivity is assumed to be a logistic function of length whilst post release mortality and illegal harvest of under sized individuals is

implemented as constant additional mortality applied to under sized fish in the catch. (Fig. B2; Table B2).

Figure B2: Selectivity (solid line) and the proportion retained (dotted line) as a function of length.



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

$$q_v^r = \exp\left(\sum_y \sum_m \ln F_{y,m,v}^r / \sum_y \sum_m \ln\{E_{y,m,v}^r (B_{y,m}^r / B_0^r)^\phi\}\right) \quad (\text{B.23})$$

where the summations over year are restricted to the years for which effort data are available (see *Historical Catch and Effort Data* below). This approach cannot be applied to reefs for which there are no catch and effort data. Thus, the commercial catchability coefficient for a reef for which there are no catch and effort data is taken to be the catchability coefficient for the closest reef. This approach is not used for the charter and recreational effort-classes because it would lead to an unrealistic spread of charter and recreational effort over the Great Barrier Reef in the future. Instead, the catchability coefficients for the charter and recreational effort-classes for reefs without catch and effort information is set equal to zero which, given the effort allocation algorithm, prevents recreational effort occurring on those reefs in the future and renders only a small amount of ‘exploratory’ fishing by the charter fleet.

Establishing Initial Conditions for Simulations

The model assumes that the population was at pre-exploitation equilibrium with the corresponding age- and sex-structure at the start of 1965. The population sizes on each reef at the start of the first year of the projection period (1999) and the corresponding age- and sex-structures are computed using the following algorithm:

- a) The number of 20cm+ animals on reef r , n^r , is generated from a lognormal distribution, $LN(I_1^r rp^r, 0.5^2)$ where rp^r is the perimeter of reef r in kilometers.
- b) The number of 1-year-olds of reef r is then determined using the formula:

$$N_{0,1}^r = n^r / \sum_{a=1}^x \sum_k \tilde{N}_a^k \quad (\text{B.24})$$

where \tilde{N}_a^k is the age-structure of the population in a virgin state, expressed as a fraction of the number of 1-year-olds and the summation over age and growth group is restricted to $L_{k,a} > 20\text{cm}$.

- c) The age- and size-structure of the population on each reef at the start of the first year in which a harvest strategy is to be applied is determined by projecting the population from pre-exploitation equilibrium to the start of 1999 with random variation in recruitment and natural mortality and catches equal to the actual estimated/assumed

catches by 6'x6' block. In order to conduct this projection, it is necessary to allocate the historical catches by block and month to reef. This is achieved using the formula⁷:

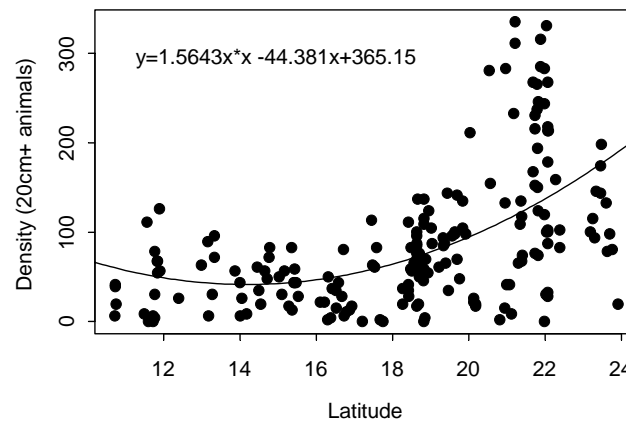
$$C_{y,m,v}^{r,b} = \frac{C_{y,m,v}^b B_{y,m}^{r,b}}{\sum_{r'} B_{y,m}^{r',b}} \quad (\text{B.25})$$

- d) The biomass corresponding to the generated value for n^r can be such that the population would be extinct at the start of the projection period. If this occurs, the value for I_1^r is increased by 5% and steps a) to d) are repeated.

The values for the reef perimeters for the reefs that have been documented are based on GBRMPA data. Historical catch and effort data are available, however, for 6' x 6' blocks in which no documented reefs are found. Of the 3,263 6' x 6' blocks that contain some effort, 2,098 do not contain any documented reefs⁸. Therefore, the model contains an option to assume that these blocks each contain a 'virtual' reef (at their centers) and to set its reef perimeter to that for the reef for which the total effort over period 1989–98 is closest to that for the 'virtual' reef. Only the 10 closest reefs to the 'virtual' reef are considered in this process to retain any spatial attributes of reef perimeter.

The base-case values assumed the I_1^r are derived from a relationship between reef density and latitude (Fig. B3). The value for I_1^r is determined by dividing the value from the curve in Figure B3 by the value for a latitude of 16.5°S (i.e. $I_1^r=1$ for reefs at 16.5°S). This divisor can be modified to achieve different scenarios regarding the status of the resource at the start of 1999.

Figure B3: Density (20cm+ animals per unit area) from visual surveys and a fitted quadratic curve.



Historical catch and effort data

The information required to initialize the operating model and to compute the catchability coefficients by reef are the catches and efforts by month, block and effort-class from 1965 to 1998 (i.e. the $C_{y,m,v}^b$ and the $E_{y,m,v}^b$). Information on commercial catch and effort is available for 1989–98 (commercial effort-class), 1996-98 (charter effort-class), and 1998 (recreational effort-class).

⁷ Equation (A.25) is an approximate solution to the system of equations defined by Equations (A.18a), (A.18b) and (A.22). This approximation has been chosen because it avoids the need to solve a large system of non-linear equations in each year.

⁸ These 2,098 blocks contain, however, only about 3% of the total effort recorded during 1989–96.

It is necessary to extrapolate this information to compute catches by month and block for 1965–88 (commercial effort-class), 1965–95 (charter effort-class), and 1965–97 (recreational effort-class). This is achieved using the following algorithm (the years listed relate to the application for the commercial effort-class):

- a) The mean catch by month and block is computed from the data for 1989–92.
- b) The catch by month for the years 1965–91 is then computed by assuming that the catch increased linearly from 0 in 1965 to the mean computed at step a) in 1991.
- c) The results obtained at step b) are assumed to apply to the years 1965–89.

This approach has been adopted because it captures the seasonality of the fishery (as reflected in the data for the years 1989–92) and because a linear increase in catch is the most parsimonious representation of the change in catch over time. A similar procedure to that described is used to generate historical catches by the charter and recreational effort-classes, except that algorithm is based on the data for 1996–98 and 1998 respectively.

Further work

- Include alternative definitions for the definition of the reproductive component of the population (mature females or males) and for the relationship between egg production and subsequent recruitment (Beverton-Holt / Ricker).
- Allow for spatial differences in the biological parameters (natural mortality, growth, fecundity, and sex-change).
- Provide model-based larval mixing matrices for the entire GBR, including all the reefs included in the model.
- The scheme used to determine the initial conditions, while conceptually what is needed, has several deficiencies. Prime among these is that the distribution for n is largely arbitrary. An attempt should (must) be made to examine the data for 'green' reefs to determine the dependence of x on longitude, latitude, location within the reef complex, etc.
- Modify the approach used to model larval dispersal to avoid overemphasizing self-seeding.

Annex B1 : The Derivation of Equation (B.8)

Consider a population at pre-exploitation equilibrium and denote the pre-exploitation value of a quantity by a subscript '0' (for example S_0^r is the (female) spawner biomass on reef r at pre-exploitation equilibrium). The number of zero-year-olds at pre-exploitation equilibrium on reef r is defined by the summation over growth group of Equation (B.4):

$$N_{0,0}^r = \sum_k K^k (ss \overset{\%}{f} S_0^r + c^r BL_0^r) = ss \overset{\%}{f} S_0^r + c^r BL_0^r \quad (B1.1)$$

From the definitions of st and ss , it follows that $st N_{0,0}^r = ss \overset{\%}{f} S_0^r$ or $N_{0,0}^r = \frac{ss}{st} \overset{\%}{f} S_0^r$.

Substituting $\frac{ss}{st} \overset{\%}{f} S_0^r$ for $N_{0,0}^r$ and $\sum_{r'} \overset{\%}{f}' S_0^{r'} \Omega^{r',r}$ for BL_0^r (see Equation (B.6)) in Equation (B1.1) yields:

$$\frac{ss}{st} \overset{\%}{f} S_0^r = ss \overset{\%}{f} S_0^r + c^r \sum_{r'} \overset{\%}{f}' S_0^{r'} \Omega^{r',r} \quad (B1.2)$$

Solving Equation (B1.2) for c^r then yields:

$$c^r = \frac{ss(1-st) \overset{\%}{f} S_0^r}{st \sum_{r'} \overset{\%}{f}' S_0^{r'} \Omega^{r',r}} \quad (B1.3)$$

Equation (B.8) follows from Equation (B1.3) after noting that S_0^r is proportional to $N_{0,1}^r$ if all the biological parameters are independent of reef.

Annex B2: The Relationship between Steepness and Parameters of Equation (B. 9a)

Steepness is defined as the ratio of the expected number of one-year-olds when the spawner biomass is reduced to 20% of the pre-exploitation level to the number of one-year-olds at pre-exploitation equilibrium. Assume (without loss of generality) that the number of one-year-olds at pre-exploitation equilibrium is 1 and that there is only a single reef (or equivalently that the level of fixing mortality is the same across all reefs and the biological parameters are also the same across all reefs). The number of one-year-olds as a function of the fully-selected fishing mortality, $R(F)$, is given by:

$$R(F) = \frac{S(F)}{S(0)} \alpha e^{-\beta U(F)/U(0)} \quad (B2.1)$$

where:

$S(F)$ is the spawner biomass when the fully-selected fishing mortality is F :

$$S(F) = R(F) \sum_{a=1}^x \sum_k f_{L_{k,a}} w_{L_{k,a}} N_a^k(F) (1 - P_{L_{k,a}}) \quad (B2.2)$$

$U(F)$ is the number of 'juveniles' as a function of F :

$$U(F) = \frac{S(F)}{S(0)} + R(F) \sum_{a=2}^J \sum_k N_a^k(F) \quad (B2.3)$$

$N_a^k(F)$ is the number of a -year-old animals in growth group k when fishing mortality is F , given that the number of one-year-olds in growth group k is K^k :

$$N_a^k(F) = \begin{cases} K^k & \text{if } a = 0 \\ N_{a-1}^k(F) e^{-M_{a-1} - S_{L_{k,a-0.5}} F} & \text{if } 0 < a < x \\ N_{x-1}^k(F) e^{-M_{x-1} - S_{L_{k,x-0.5}} F} / (1 - e^{-M_x - S_{L_{k,x+0.5}} F}) & \text{if } a = x \end{cases} \quad (B2.4)$$

α , β are the parameters of the stock-recruitment relationship.

The use of a Ricker-like relationship for the mortality between ages 0 and 1 is based on the assumption that this mortality is due to competition between settling animals and the 1+ population already on the reef.

Now, evaluating Equation (B2.1) at the pre-exploitation level yields:

$$1 = \alpha e^{-\beta} \quad \text{or} \quad \alpha = e^{\beta} \quad (\text{B2.5})$$

Substituting Equation (B2.5) into Equation (B2.1) then yields:

$$R(F) = \frac{S(F)}{S(0)} e^{-\beta(U(F)/U(0)-1)} \quad (\text{B2.6})$$

Denoting $S(F)/R(F)$ as $S^{\circ}(F)$ and $U(F)/R(F)$ as $U^{\circ}(F)$, it is possible to solve Equation (B2.6) for $R(F)$:

$$R(F) = 1n[S^{\circ}(F) e^{\beta} / S^{\circ}(0)] / [\beta U^{\circ}(F) / U^{\circ}(0)] \quad (\text{B2.7})$$

The algorithm used to find the value for β (and hence through Equation (B2.5) the value of α) is:

- a) Guess a value for β and calculate the value for α from Equation (B2.5).
- b) Find the value for F such that the ratio $R(F) S^{\circ}(F) / S(0) = 0.2$ - a bisection method is used for this purpose.
- c) Compare $R(F)$ with the pre-specified value for steepness.
- d) Repeat steps a) – c) until $R(F)$ equals the pre-specified steepness.

Equation (B.9a) is then obtained from Equation (B2.6) after replacing the ratio $S(F)/S(0)$ by the product of the spawner biomass and the survival from age 0 to age 1 at pre-exploitation equilibrium.

Annex B3: Estimating the Parameters of the Growth Curves

The data on length-at-age collected during the ELF experiment (Mapstone *et al.* 1998, Davies *et al.* 1998) form the basis for the estimation of the parameters of the growth function and those that determine the fraction of 0-year-olds in each growth group. It is commonly assumed when fitting growth curves that the data on length-at-age is a random sample from the population or, at least, that the sample for a given age is a random sample from the fish of that age. However, this assumption is invalid for the length-at-age data for coral trout because, due to the selectivity of gear employed in the fishery, animals of different lengths are not equally vulnerable to capture (see Fig. B2). Ignoring gear selectivity when fitting a von Bertalanffy curve leads to negatively biased estimate of κ . Dow (1992) developed a method that, given the relationship between length and the probability of capture, corrects for this bias⁹.

Dow (1992) noted that the probability of capturing an animal of length λ is the product of the probability that it has length λ , $P(\lambda)$, and the probability that it has been captured given that it has length λ , $V(\lambda)$. The functional form of V and the values for its parameters are assumed to be known exactly (see Fig. B2). Therefore, it is only the functional form of P and the values for its parameters that have to be determined from the data. The likelihood function for a single datum is of the form:

$$L(x | \underline{\phi}) = \frac{V(x) P(x | \underline{\phi})}{\int V(y) P(y | \underline{\phi}) dy} \quad (\text{B3.1})$$

⁹ Essentially the same method was developed independently by Paul *et al.* (2000).

Notice that the denominator is a scaling constant, such that $\int L(x|\underline{\phi})dx = 1$. The total likelihood to be maximised to estimate the parameters of the growth model is the product of Equation (B3.1) over all data points. There are several ways of modelling data on length-at-age. However, for the purposes of this study, it is assumed that length-at-age is normally distributed about its expected value:

$$P(l | a, \underline{\phi}) = \sum_k \frac{K^k}{\sqrt{2\pi\sigma}} e^{-\frac{(l - \hat{L}_{k,a})^2}{2\sigma^2}} \quad (\text{B3.2})$$

where:

$\hat{L}_{k,a}$ is the model-estimate of the length of an animal of age a in growth group k (determined from a von Bertalanffy growth equation where the values of l_∞ , κ and t_0 are assumed to differ among growth groups):

$$\hat{L}_{k,a} = l_\infty^k (1 - e^{-\kappa^k (a - t_0^k)}) \quad (\text{B3.3})$$

K^k is the proportion of the population in growth group k .

Parameteric forms are assumed for l_∞ , κ , t_0 and K to reduce the number of estimable parameters:

$$\kappa^k = \bar{\kappa} - \kappa_s (k - (n + 1) / 2) \quad (\text{B3.4a})$$

$$l_\infty^k = \bar{l}_\infty - l_s (k - (n + 1) / 2) \quad (\text{B3.4b})$$

$$t_0^k = \bar{t}_0 - t_s (k - (n + 1) / 2) \quad (\text{B3.4c})$$

$$K^k = \exp(-(k - (n + 1) / 2)^2 / \sigma_K^2) \quad (\text{B3.4d})$$

where:

n is the number of growth groups (10),

\bar{l}_∞ , $\bar{\kappa}$, \bar{t}_0 are the means of the distributions for l_∞ , κ and t_0 ,

l_s , κ_s , t_s are the parameters that determine the distribution of l_∞ , κ and t_0 among growth groups, and

σ_K determines the proportion of the population in each growth group.

The data analysed consisted of 5,041 age-length data pairs. Only ages obtained after sectioning the otoliths were included in the data set and any animals for which both age and length were not available were excluded. No account was taken of the time of year when the otoliths were sampled. The model was fitted by first fixing the values for \bar{l}_∞ , $\bar{\kappa}$, and \bar{t}_0 to the values obtained from an analysis based on a single growth group and then finding the values for the parameters l_s , κ_s , t_s and σ_K that maximise the likelihood function (Table B3.1). The value for the parameter σ in Equation B3.2 was fixed at 1.5cm because the operating model does not consider errors in measuring length, only that different animals grow according to different growth curves.

Figure B3.1 shows the fits to the data, Figure B3.2(a) shows the ten growth curves and Figure B3.2(b) the weight assigned to each of these growth curves (growth curve 1 is the top growth curve in Figure B3.2(a), growth curve 2 the second growth curve from the top...). The ten growth curves cover a wide range. However, the four central curves are assigned by far the greatest weight (Figure B3.2b; Table B3.2).

Table B3.1: Estimates (with asymptotic standard errors in parenthesis) of the parameters of the von Bertalanffy growth equation.

Parameter	Value (SE)
\bar{L}_∞	563.0 (9.8)
\bar{K}	0.206 (0.0073)
\bar{t}_0	-0.881 (0.204)
l_s	40.31 (2.36)
K_s	0.0219 (0.0241)
t_s	-2.24 (0.411)
σ_K	2.158 (0.0039)

Figure B3.1: The fits to the ageing data. The solid dots indicate the actual data, the dotted bars the population length-at-age distributions, the solid line the selectivity pattern, and the solid bars the catch length-at-age distributions.

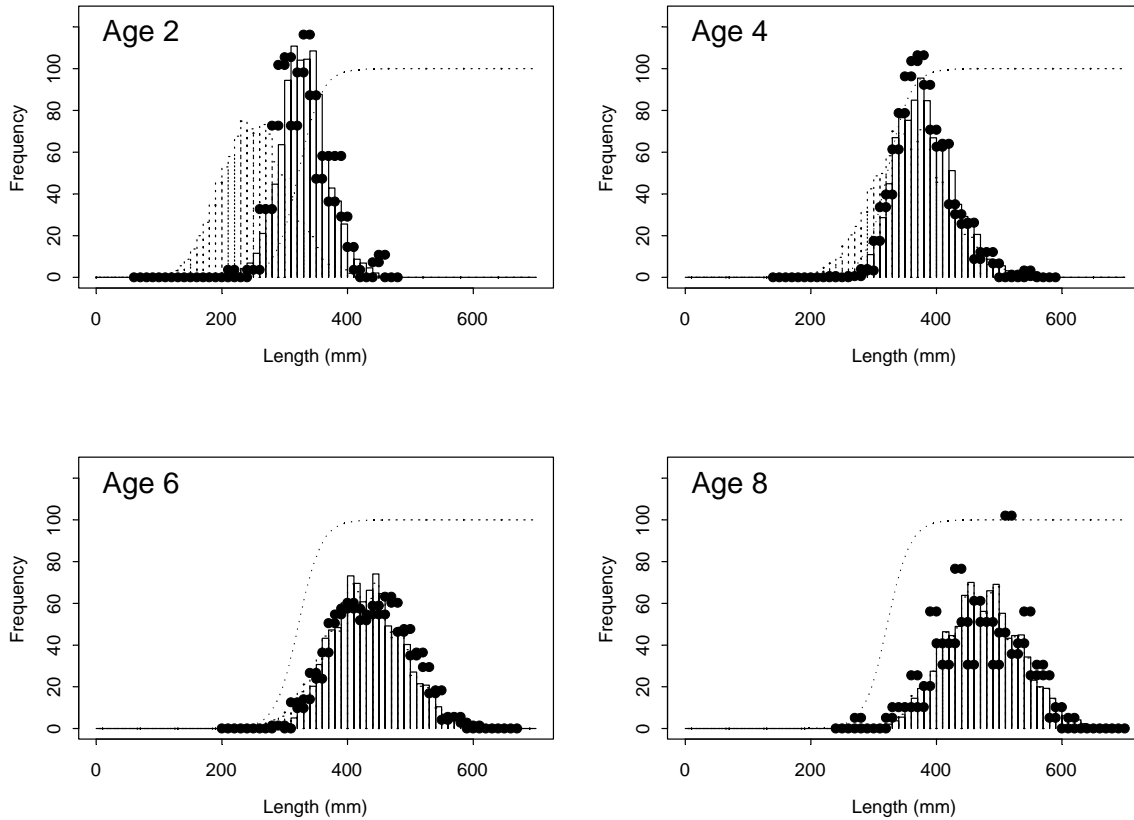


Figure B3.2: Growth curves by growth group (left panel) and the relative weight assigned to each growth group (right panel).

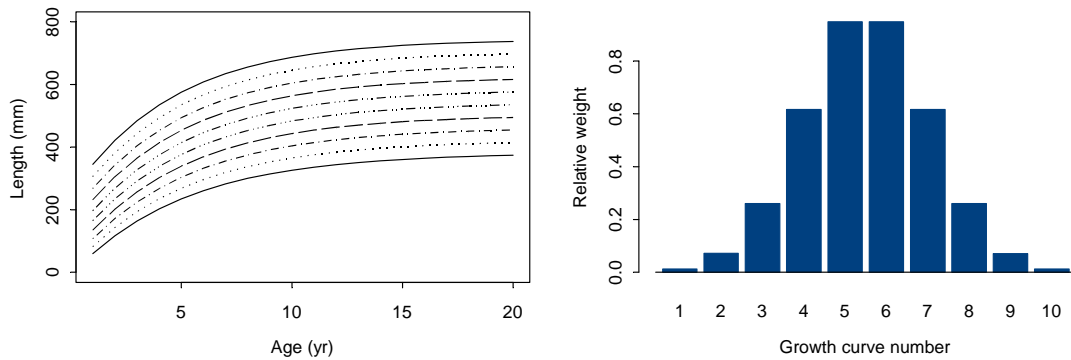


Table B3.2: The growth parameters for each growth group and the relative proportion of births in each growth group.

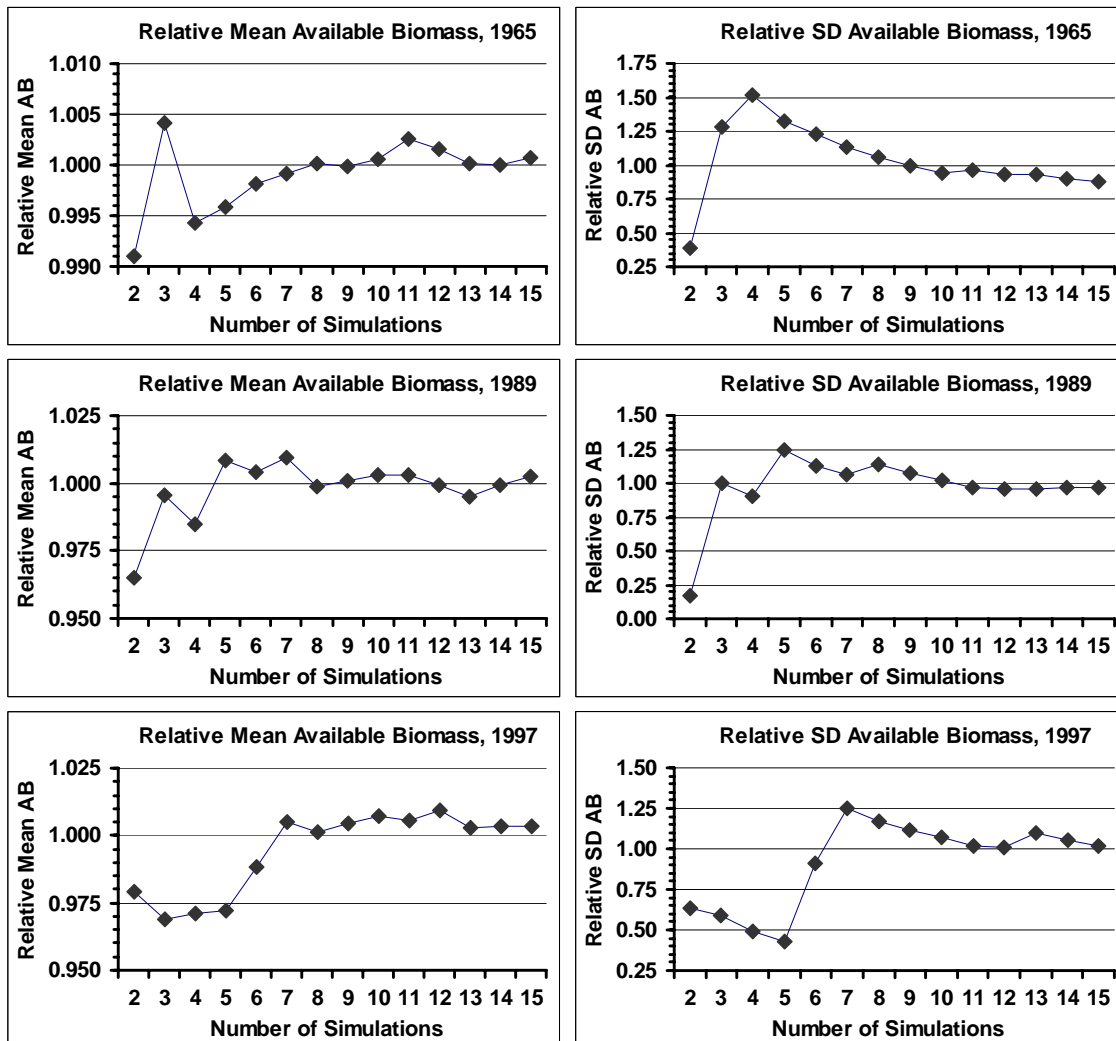
Growth Group	Parameter Values			
	l_{∞} (mm)	κ	t_0 (yr)	K
1	744.4	0.216	-1.889	0.003
2	704.1	0.214	-1.665	0.019
3	663.8	0.211	-1.441	0.068
4	623.4	0.209	-1.217	0.161
5	583.1	0.207	-0.993	0.248
6	542.8	0.205	-0.769	0.248
7	502.5	0.203	-0.545	0.161
8	462.2	0.200	-0.320	0.068
9	421.9	0.198	-0.096	0.019
10	381.6	0.196	0.128	0.003

Appendix C: Summary Diagnostics and Sensitivity Analyses for ELFSim

In this appendix we present summaries of some of the diagnostics and sensitivity analyses done during the development and application of ELFSim. The material presented is not exhaustive but is intended to illustrate the consequences of key assumptions and uncertainties about which we had to make decisions prior to running the MSEs presented in the body of the report. In particular, our emphasis is on whether the results of comparisons of alternative strategy sets would be qualitatively different under alternative assumptions or default model settings. We use outputs for available biomass to illustrate all points, but results for other variable were qualitatively similar to those for available biomass.

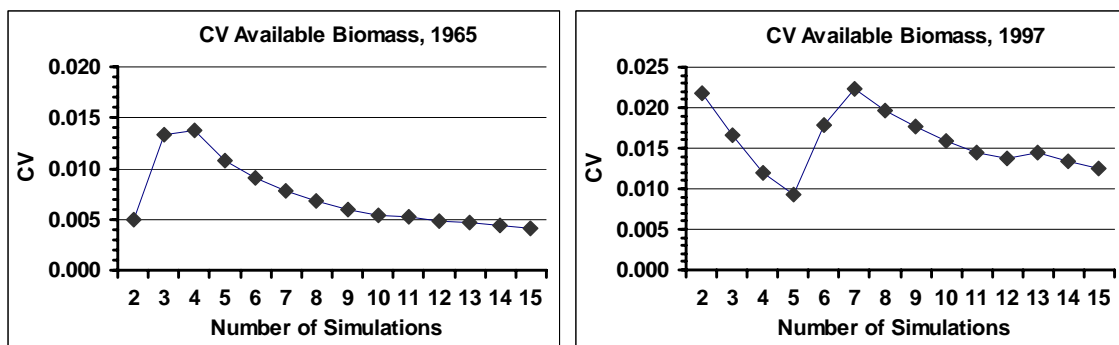
Required Numbers of Simulations

Figure C1: Mean available biomass from different numbers of simulations under the status quo scenario as a proportion of the grand mean of all simulations (Left) and mean standard deviation (SD) as a proportion of grand mean standard deviation from the same runs (Right) plotted against number of simulations. Plots are for estimated virgin available biomass (1965, Top), available biomass in the first year of real catch and effort data, after hind-casting (1989, Middle) and in the penultimate year before projections (1997, Bottom).



Estimates of mean values of performance indicators stabilised rapidly with increasing numbers of simulations for a given scenario, as did variation among replicate runs of the same scenario (Fig. C1). After only 8-10 runs, standard deviations of the estimates were consistently very small relative to the average values of performance indicators (CV ~1-3%) and showed very slow further improvement with additional runs (Fig. C2). Thus, there was little advantage to running more than 10 replicate projections of each strategy set. The low variation among sets arose because performance indicators were derived from data summed over hundreds or thousands of reefs. Whilst variation in most variables among reefs and for individual reefs through time was relatively large, that reef-specific variation was not correlated over the entire domain, meaning that the summed results tended to be relatively invariant.

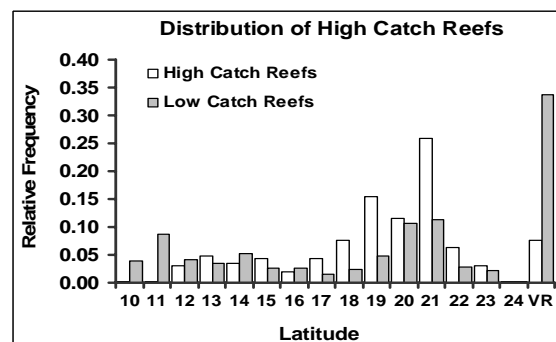
Figure C2: Coefficient of Variation ($CV = SE/Mean$) from different numbers of simulations under the status quo scenario plotted against number of simulations for estimated virgin available biomass (1965, Left) and available biomass in the penultimate year before projections (1997, Right).



Distribution of Catch and Effort

Catch and effort by all sectors was non-uniform over the GBR, as described previously by Mapstone *et al.* (1996a) and QFS (2002), Green *et al.* (in prep.) and Higgs (1996, 1999). Approximately 75% of all reported commercial catch between 1989 and 1998 was taken from just 658 or 17.2% of the possible 3822 reefs (including virtual reefs) in the simulation domain (the GBR Marine Park). These 658 'high catch' reefs were non-uniformly distributed and also distributed differently along the GBR than the remaining 3,164 reefs, with high catch reefs accounting for disproportionately more of the reefs between 17°S and 21°S than the low catch reefs. The high catch reefs also accounted for disproportionately fewer of the virtual reefs and reefs in the far north of the GBR Marine Park (Fig. C3). This distribution reflects the previously documented distribution of fishing effort in the GBR, with catch and effort being greatest between Cairns (17°S) and the Swains Reefs (22°S).

Figure C3: Relative frequencies of 'high catch reefs' and the remaining 'low catch reefs' by latitude along the GBR. Also shown is the proportion of each group that were virtual reefs (VR).



The high catch reefs accounted for ~67% of mapped reef area and ~35% of reef perimeters, indicating that they tended to be larger than other reefs. High catch reefs also are under-represented compared to other reefs in the current closure scenario (Table C1). This inequity is largely attributable to the disproportionate number of closed reefs in the northern GBR under existing zoning strategies and was corrected somewhat under the increased closure regimes considered for this report (Table C1) as increasing closures were applied in the southern GBR. Despite this ‘balancing up’, the proportions of previous catch and effort from reefs that were open or closed in future scenarios represented by that from high catch reefs was relatively constant across all scenarios (Table C2). This indicates that the impacts of alternative closure scenarios on the re-distribution of effort and consequent catch was not disproportionately biased in relation to reefs that previously had experienced high or low catch and effort (Table C2). The high catch reefs were defined in terms of the commercial catch from them and they accounted for relatively smaller proportions of recreational catch and effort historically. This is a consequences of the biased distribution of recreational effort toward near shore sites and commercial and charter effort toward off-shore reefs.

Effects of Habitat Scalar and Uneven Catch-Effort Distribution

The relative concentration of effort and catch on the high catch reefs meant that those reefs showed slightly exaggerated responses to management scenarios (such as closures) compared with the responses summed over all reefs (Fig. C4). Similarly, reducing the value of the Habitat Scalar increased the contrast among alternative management scenarios for all performance indicators, though the relationships among different strategy sets were qualitatively the same irrespective of chosen value for Habitat Scalar. The difference between high catch reefs and other reefs tended to be more accentuated as Habitat Scalar was increased (Fig. C4).

Figure C4: Mean available biomass over the last five years of the projection period (2021-25) as a proportion of virgin available biomass on all closed reefs (Left) and on historically high catch reefs (**HC Reefs**) that were closed to fishing (Right) under four regimes of fishing effort (**0, 0.5, 1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current ≈ 16%) when the Habitat Scalar (**HS**) was set at 0.25 (Top) and 0.5 (Bottom). Error bars are Standard Errors.

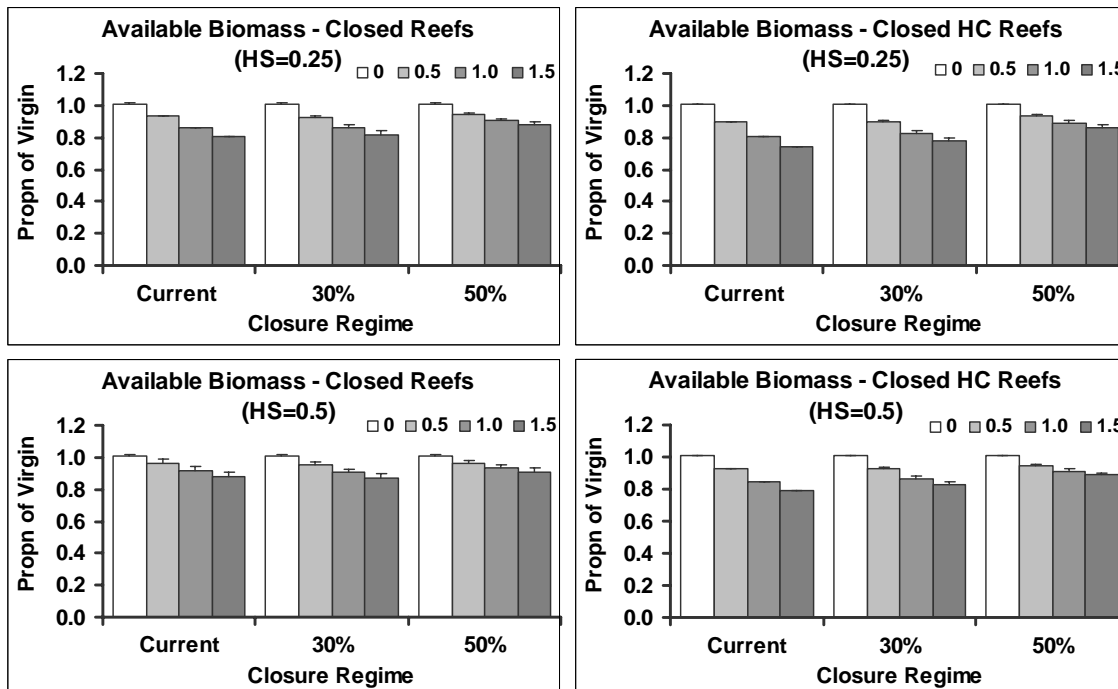


Table C1: Representation of 'high catch' reefs by frequency (**N°**), area (**km²**) and perimeter (**km**) **Open** or **Closed** to fishing under each **Closure Regime**. Also shown are the respective percentages of all reef habitats in each closure regime represented by high catch reefs (**% Hab**), the percentage of the high catch reefs in each open or closed scenario (**% HC**), and the percentage change from the current closure regime (**%Cc**) incurred as other closure regimes affected high catch reefs.

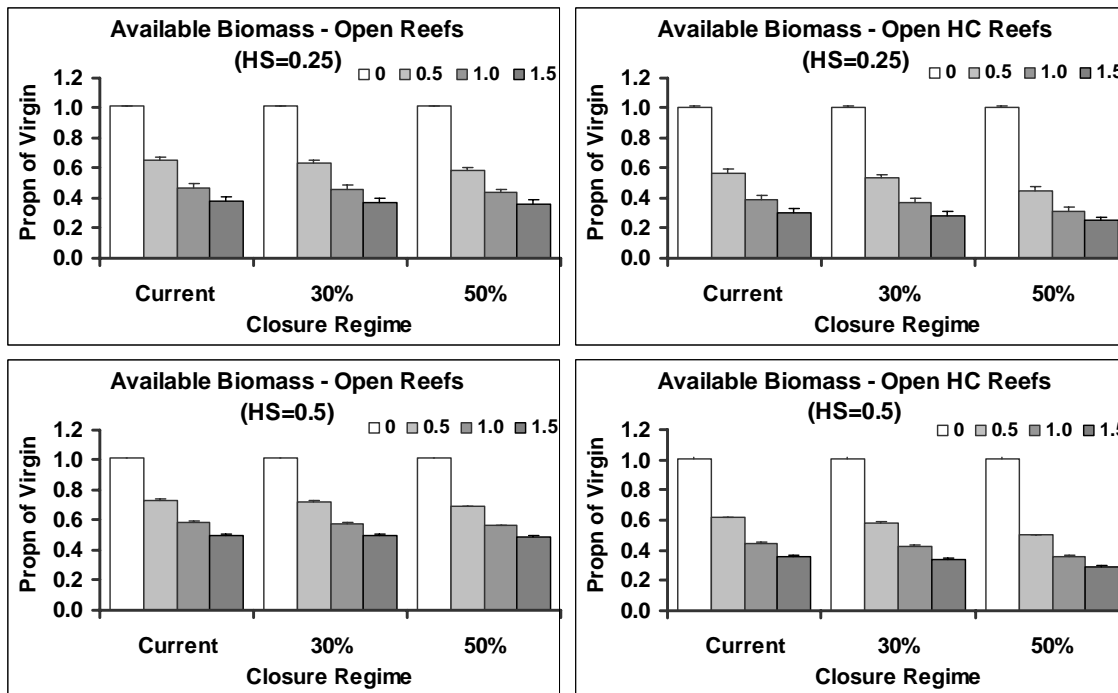
Closure		N° Reefs				Reef Area				Reef Perimeter			
Status	Regime	N°	% Hab	% HC	% Cc	km ²	% Hab	% HC	%Cc	km	% Hab	% HC	%Cc
Open	0%	658	17.2	100		13673	67.1	100		12524	34.8	100	
	Current	603	18.6	91.6		12279	77.3	89.8		11263	37.3	89.9	
	30%	505	17.3	76.7	-16.3	9934	78.8	72.7	-19.1	9177	34.4	73.7	-18.52
	50%	372	15.5	56.5	-38.3	6391	75.1	46.7	-48.0	6319	29.3	50.5	-43.9
Closed	0%	0		0		0		0		0		0	
	Current	55	9.5	8.4		1393	31	10.2		1261	21.7	10.1	
	30%	153	16.9	23.3	178.2	3738	48.1	27.3	168.3	3347	35.7	26.7	165.43
	50%	286	20	43.5	420.0	7282	61.3	53.3	422.8	6208	42.9	49.5	392.3

Table C2: Mean reported **Catch** (t) and **Effort** (line-days) by Commercial (1989-98), Charter (1996-98) and Recreational (1998) fishing fleets from the 'high catch' reefs destined to be **Open** or **Closed** in each of the **Closure Regimes**. Also shown are the respective percentages of all reported catch (**%Cat**) and effort (**%Eff**) by each fleet that had occurred on high catch reefs subsequently included in each closure regime.

Closure		Commercial Fleet				Charter Fleet				Recreational Fleet			
Status	Regime	Catch	Effort	%Cat	%Eff	Catch	Effort	%Cat	%Eff	Catch	Effort	%Cat	%Eff
Open	0%	1,081.4	47,275	77.4	76.9	34.0	18,113.	67.3	68.5	61.7	31,519.	17.9	17.9
	Current	1,017.8	44,367	78.4	77.8	30.7	16,459.	69.8	71.5	56.8	29,010.	17.7	17.7
	30%	801.7	35,169	76.1	75.2	19.9	11,849.	64.8	67.4	52.1	26,646.	16.9	16.9
	50%	554.4	24,411	74.1	72.7	17.2	10,476.	67.7	68.6	40.5	20,688.	14.9	14.9
Closed	0%	0.0	0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Current	63.6	2,907	64.6	65.4	3.2	1,654.5	50.2	48.4	4.9	2,509.3	20.0	20.0
	30%	279.7	12,106	81.5	82.1	14.1	6,263.8	71.3	70.7	9.5	4,873.5	25.6	25.6
	50%	527.0	22,864	81.2	81.9	16.8	7,637.3	67.0	68.5	21.2	10,831.	28.7	28.7

Similar results were obtained for open reefs, with contrasts among strategy sets being greater for high catch reefs than for all reefs and contrasts being diminished with higher values for the Habitat Scalar (Fig. C5). Importantly, however, neither the variations in Habitat Scalar we considered nor considering the uneven distribution of fishing catch and effort changed qualitatively comparisons of alternative strategy sets.

Figure C5: Mean available biomass over the last five years of the projection period (2021-25) as a proportion of virgin available biomass on all reefs open to fishing (Left) and historically high catch reefs (**HC Reefs**) open to fishing (Right) under four regimes of fishing effort (**0, 0.5, 1.0** and **1.5** x 1996 level) and three regimes of area closure (% of reef perimeters; current \approx 16%) when the Habitat Scalar (**HS**) was set at 0.25 (Top) and 0.5 (Bottom). Error bars are Standard Errors.



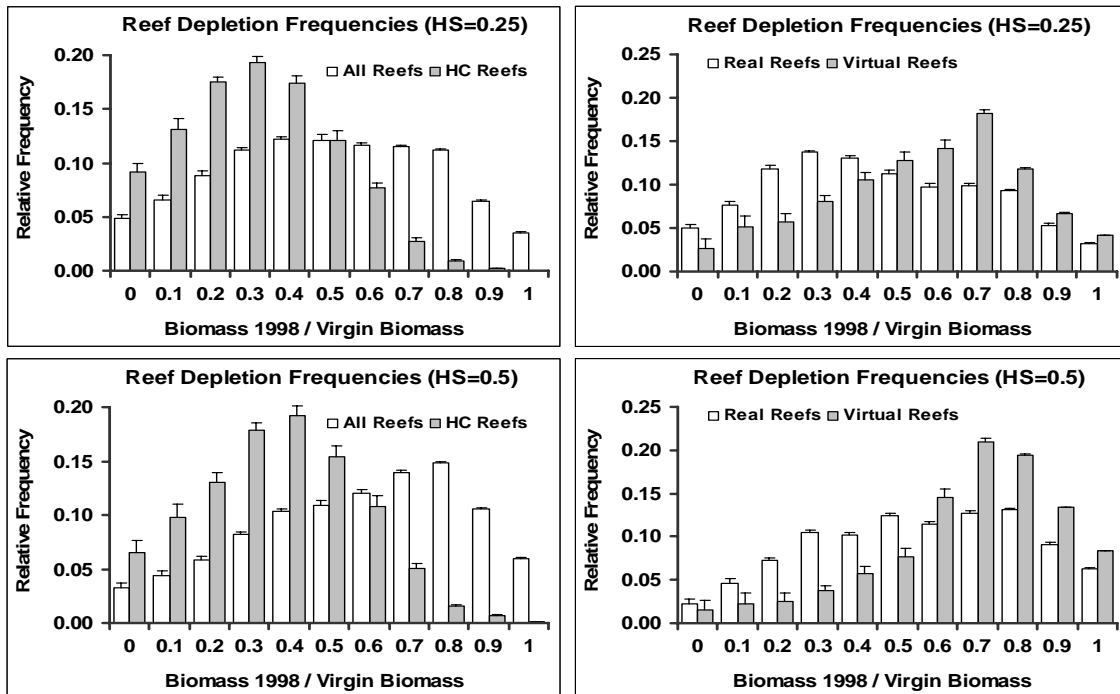
Relative Depletion of Reefs

The value of the Habitat Scalar also influenced the relative depletion of reef-specific biomass during the intialisation period (Fig. C6). This result is to be expected since the Habitat Scalar basically scaled the size of the resource available for harvest, with the amount of harvest during the historical period coming from data and thus remaining constant for all simulations. As the value of the Habitat Scalar increased, the distribution of reefs according to their depletion by historical fishing shifted to the right, with more reefs remaining closer to virgin status than under lower values of the Habitat Scalar (Fig. C6). This effect was less evident for high catch reefs and virtual reefs than when all reefs or all gazetted reefs were considered (Fig. C6).

High catch reefs tended to be more severely depleted by historical fishing irrespective of the value of the Habitat Scalar, as might be expected by virtue of the concentration of fishing on them. Virtual reefs, however, generally were less depleted than other reefs. This effect arose because catches from virtual reefs tended to be small compared to those from other reefs and because the process for allocating perimeter to them meant that the amount of habitat represented by them was not different from other reefs. Thus, the resources on virtual reefs remained relatively lightly exploited. Since effort allocation to these reefs during

future projections was constrained to remain within the bounds of historical effort, this situation would have been perpetuated in projections. This would likely have attenuated the aggregate effects of fishing on open reefs.

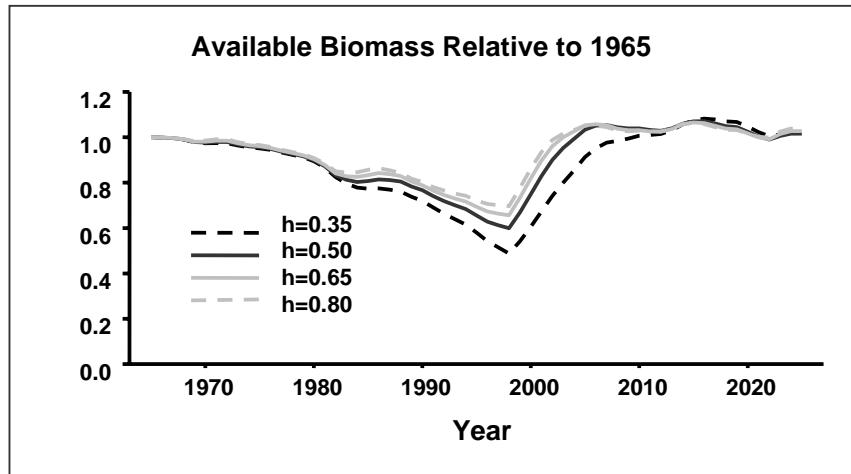
Figure C6: Distribution of reef-specific available biomass at the end of the initialisation period (1998) as a proportion of virgin (unfished) available biomass for all reefs compared with high catch reefs (Left) and gazetted and virtual reefs (Right) when the Habitat Scalar (**HS**) was set at 0.25 (Top) and 0.5 (Bottom). Error bars are Standard Errors.



Effects of Different Values for the Steepness Parameter

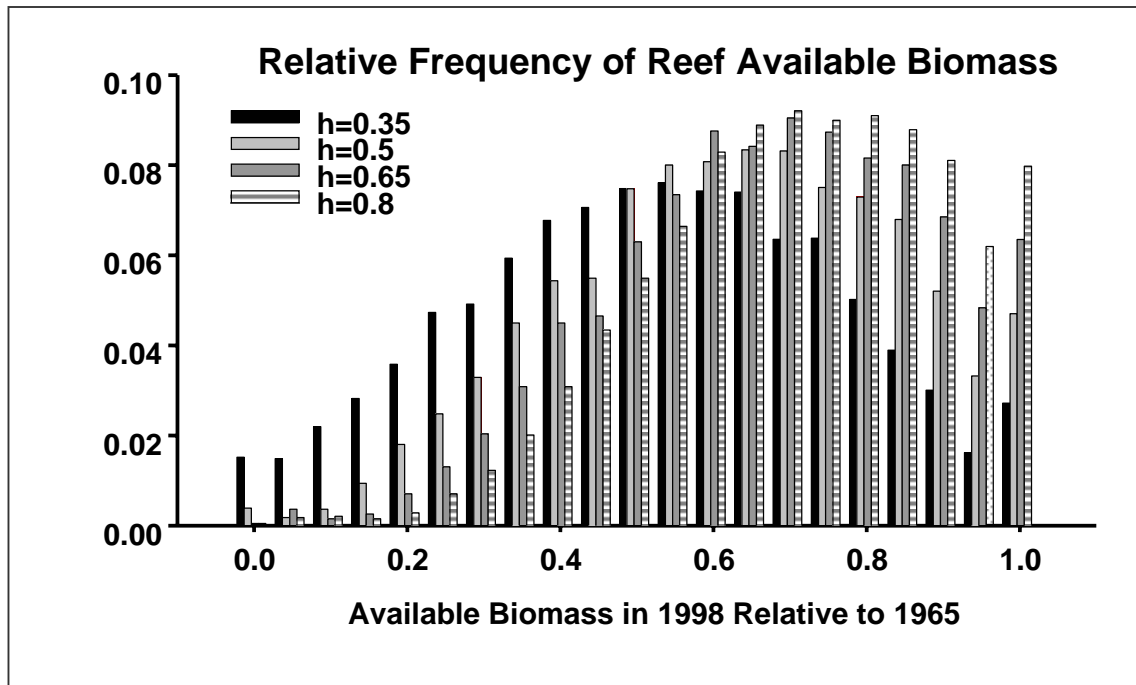
The steepness parameter (Francis 1992) is a measure of the expected recruitment at some defined stage (in our case, at aged 1) when the biomass of spawners is reduced to 20% of its virgin level. Thus, higher values for steepness will equate with higher likelihood of large recruitment when biomass is depleted than when steepness is lower. This effectively means that populations modelled with higher steepness values are more buffered against recruitment-mediated impacts of harvest than those with lower steepness (Fig. C7). In the absence of an empirical stock-recruitment relationship for coral trout, we considered 4 values of steepness in terms of their impacts on the ability of stocks to rebuild when released from fishing pressure at the beginning of the projection period (Fig. C7). In all cases, populations had stabilised after 5-10 years of projections, with the lowest value for steepness precipitating a significantly greater depletion status after initialization and a substantially greater lag in recovery than the three larger values of steepness.

Figure C7: Trajectories of total available biomass as a proportion of virgin available biomass (1965) under four different values of steepness (Francis 1992). Responses during the projection period are those expected when fishing stopped at the end of the initialisation period (1998).

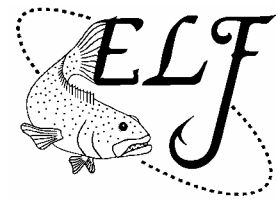


Changing the steepness parameter affected the frequency distribution of reef-specific depletion, with higher steepness values reducing the degree of depletion of reefs by historical fishing (Fig. C8). The most dramatic shift in depletions occurred with steepness being decreased from 0.5 to 0.35 (Fig. C8). We used a steepness value of 0.5 in our simulations because it produced results fairly similar to the higher values, but intermediate between the extremes (Fig. C7, Fig. C8).

Figure C8: Relative frequencies of reef-specific available biomass at the end of initialisations relative to virgin available biomass (1965) for four values of steepness.



Appendix D: Report from the ELF MSE 1st Stakeholder Workshop



REPORT FROM

EFFECTS OF LINE FISHING

PROJECT MANAGEMENT

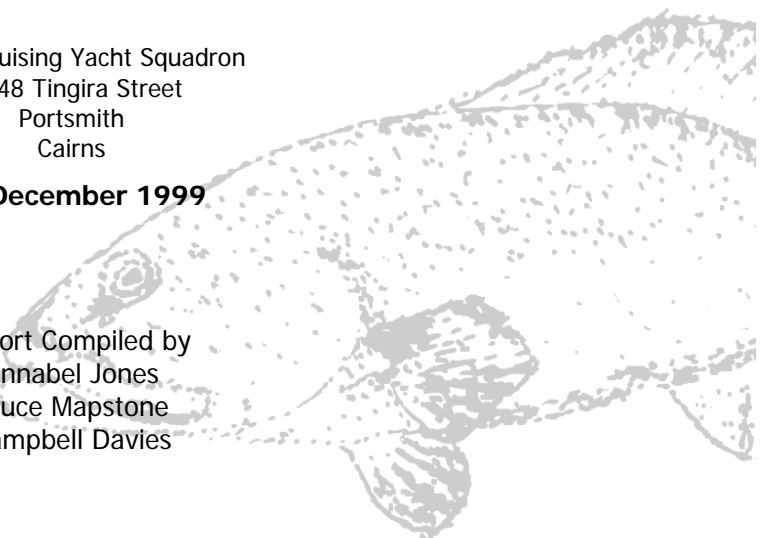
STRATEGY EVALUATION

1ST STAKEHOLDERS WORKSHOP

Cairns Cruising Yacht Squadron
42-48 Tingira Street
Portsmith
Cairns

1st December 1999

Report Compiled by
Annabel Jones
Bruce Mapstone
Campbell Davies





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Introduction

December 1st, 1999 was a landmark day for the Effects of Line Fishing Project heralding a major milestone for the project. The event was the first of a series of meetings with stakeholders to initiate discussions of one of the vehicles for transferring the outcomes from the ELF Project into management policy for the Reef Line Fishery. This is being developed specifically for the Queensland Reef Line Fishery by the ELF Research Team and will provide essential tools for managers of the fishery, and those groups that advise them. These tools are referred to as Management Strategy Evaluation, or MSE. Specifically, MSE involves using computer models to evaluate the relative performance of different management strategies in meeting management objectives for the Reef Line Fishery.

This document summarises the information that was provided to the stakeholder representatives at this meeting and the discussions that occurred. It provides a background to the MSE approach, what MSE is, and what it can and can't do, how MSE fits into the ELF Project and how the ELF Experiment is providing data for the development of the MSE models. The slides presented at the workshop accompany this document as an attachment.

The ELF team welcomes any comments regarding this document, the first ELF MSE Stakeholders Workshop, the MSE approach, or any other aspect of the ELF Project. Questions or comments can be directed to:

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The ELF Project

The ELF Project is based at the CRC for the Great Barrier Reef World Heritage Area (CRC Reef). The research team comprises a collaborative team of researchers from James Cook University (JCU), the Australian Institute of Marine Science (AIMS), the Queensland Department of Primary Industries (QDPI), and the CSIRO Marine Division. In addition, the project has developed strong links with representative industry bodies such as the Queensland Commercial Fisherman's Organisation (QCFO), Sunfish, and fishers from all sectors of the fishery, and the main management agencies (QFMA and GBRMPA).

The ELF Project consists of four core areas of research:

- 1) Fisheries information and fleet characteristics
- 2) Biology of target and non-target species
- 3) Modeling and Management Strategy Evaluation
- 4) Liaison and Information Transfer

Issues for Management of the Reef Line Fishery

The GBR region supports multiple use and is a World Heritage listed area of ecological significance. Management of the region, therefore, must facilitate continued multiple use of the GBR (including fishing), whilst ensuring that such use does not threaten the ecological, aesthetic and cultural values for which it was listed.

The fishery in the GBR Region is accessed by a variety of user groups including recreational fishers, charter boat operators, commercial fishers and non-fishers. These users have access to the same areas, but with some differences in management régimes applied depending on activity. The different sectors have very different motivations and behavior, and consequently different distributions of effort.

The Queensland Reef Line Fishery is a complex one in that it is a multi-species fishery, where a large number of fish species are targeted by fishers in most areas. However, coral trout and red throat emperor are the major target species and comprise the majority of the catch. The biology of many tropical reef species, including the main target species can also complicate management of the fishery. Many have complex life history strategies (for example many change sex during their life), and their abundance varies from one region of the GBR to another.

Effective management of fishing requires knowledge of the characteristics of the fisheries (catch, effort, fishing practices, economics etc) and of the consequences of fishing for the harvested stocks and the ecosystem. Management of line fishing to date has been hampered, however, by a shortage of information about fishers' activities, their effect on the resource and our knowledge of the biology and dynamics of fish species.

The ELF Project was established to provide the necessary technical base for quantitative evaluations of potential management strategies for line fishing in the GBR environment.

What is Management Strategy Evaluation?

Management Strategy Evaluation (MSE) involves assessing the strengths and weaknesses of different management strategies by showing the likely consequences of each if implemented. Following evaluation, the results are presented in a way which lays bare the trade-offs in performance of each strategy across a range of management objectives, rather than trying to identify a single 'best option'.

A major advantage of the MSE approach over other methods is that uncertainty in what is known about the status of the fishery and its response to fishing is taken into account, incorporated as a measure of risk of a strategy not meeting particular objectives. Thus the risk involved with each strategy is clear to the stakeholders.

The MSE approach provides a transparent way for stakeholders to be involved in the refinement of measurable objectives and alternative strategies, and the evaluation process. This approach can significantly enhance the level of understanding of key areas of uncertainty in the assessment and management of the fishery. This approach will also give a sense of ownership in fisheries management among stakeholders, hopefully resulting in greater levels of agreement on final management decisions.

The MSE approach requires a number of key ingredients. These are:

- A clearly defined set of quantitative management objectives (termed operational objectives)
- A set of criteria to measure the performance of each strategy in meeting the objectives
- A means of calculating these performance criteria for each management strategy
- A set of management strategies to be considered

An example of a general management objective could be, 'to maintain a stable level of catch'. However a more measurable version of this example would be 'to maintain catches such that there is a very low risk of more than 25% change in catch from one year to another'.

Performance measures are an indication of how well a particular objective is met by different strategies. A performance measure for the example given above would be the average change in catch between years over the period of the evaluation (e.g. 20-30 years).

A number of strategies could be implemented to meet the objective given as an example above. One such strategy could be a reduction in the number of operating boats (effort controls). Another strategy could be the implementation of a catch quota (harvest control) and yet another strategy may be implementation of area closures to fishing.

A range of each or all of the strategies can be evaluated. For example for effort controls, the evaluations may include cutting effort by 10%, 20% or 30%. Which one would provide the greatest chance of meeting the above management objective as indicated by the performance measure? This is the job of the MSE, in that it provides an objective way to compare alternatives and include major sources of uncertainty.

The First MSE Stakeholders Workshop

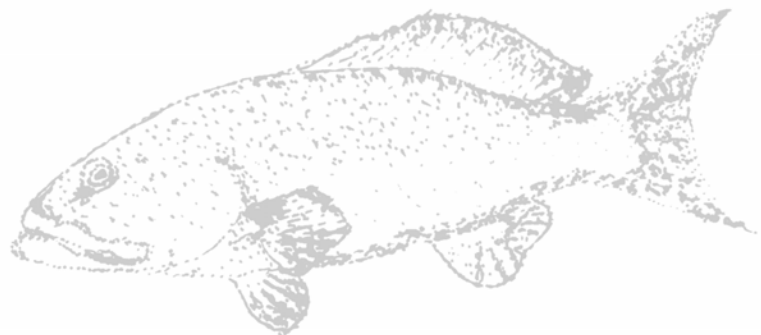
On the surface, this may have appeared as simply a meeting bringing together various representatives and their relevant expertise, from a range of stakeholder groups with a collective interest in future management of the Queensland Line Fishing Resource. However, the relevance of this meeting went much deeper.

The aims of this meeting were two fold: Firstly, to inform stakeholders about how MSE's work and how they will be of benefit to them. Secondly and most importantly, the meeting aimed at fulfilling one of the most important requirements in the development of MSE's - refining a set of quantitative management objectives, performance measures and alternative strategies that stakeholders may ask to be evaluated once the MSE models are operational. This is a crucial step in MSE development to correctly 'set the boundaries' for the MSE models. However, this step is often a major hurdle in the development of these tools.

Workshop Goals

"To let you know where we are up to and where we are going with ELF tools for Management Strategy Evaluation (MSE), and to get your help to focus the ELF tools on things that matter to you in the reef line fishery."

1. To familiarise stakeholders with current work in MSE arising from the ELF Project to aid management of the GBR Reef Line Fishery -
 - 1.1. What the ELF Project can/cannot provide.
2. To initiate formal discussions with stakeholders on:
 - 2.1. **Specific** operational objectives for future management of the fishery.
 - 2.2. Harvest/Conservation management strategies and performance indicators for each strategy.
 - 2.3. Performance measures for use in MSE for the Reef Line Fishery.
3. To discuss with stakeholders their expectations of the MSE approach.
4. To clarify data requirements for further development of ELF MSE.



Meeting Summary

Opening Comments

- John Kerin and Bruce Mapstone

It is now clear that it is only through research, education and science that natural resource management will work efficiently and be successful. This is reflected in the fact that ReefMac has been working for four years to get agreement on the Coral Reef Fin Fish Management Plan. The debate on this plan continues, and it is clear that the tools being developed by the ELF Team would be helpful in this matter. However, this workshop was not aimed at debating current policy or management plans, but to discuss the ELF Management Strategy Evaluation tools, their use and implementation.

General Background to Quantitative Fisheries Management

- Tony Smith

A fishery is a complex resource where the system is constantly in a state of flux. In a dynamic system such as this, management will always be a challenge - What is a sustainable level of fishing in the long term? - What is the best strategy for managing the fish stocks to meet the needs of multiple users, often with conflicting needs? This is the challenge for fisheries management in Queensland.

This workshop is an important part of a longer term goal aimed to provide tools (the ELF MSE model) and information to stakeholders, and in particular the ReefMAC, to assist them in making management decisions for the Queensland Reef Line Fishery. This process will aid identification of longer-term strategies for managing the fishery.

In the first instance the focus of the ELF Project MSE will be on management of common coral trout (*Plectropomus leopardus*) as the main target species in this fishery. However, the longer term plan will be to produce models that are applicable to a multi-species fishery such as the GBR Line Fishery.

A key step in the process is to refine specific, measurable management objectives from the more general ones that stakeholders have for management of the fishery. Management strategies are put in place to deliver the management objectives. This is where the ELF management strategy evaluation models become important. These tools will help decision makers such as ReefMac, by predicting the probability that particular strategies will achieve various management objectives, clearly identifying trade-offs between strategies. The decision-makers can then make an informed decision on which strategy is most likely to meet a particular set of objectives, given current knowledge of the system.

The benefits of using a MSE approach, such as being developed by the ELF team, is that it allows 'pre-testing' of management strategies before putting them into legislation.

- Issues raised from the floor

It is important to include flexibility in the models to take into account any uncertainty in the fishery. The Fishery can not be molded to fit the model.

Response: This is an important point, and it is in this area that the MSE approach has some advantage. By identifying areas of uncertainty in the resource - for example, the variation in recruitment levels of young fish to the fishery each year (something we know little about) - this uncertainty can be incorporated into MSE models and taken into account when examining the strengths and weaknesses of various potential management strategies. A major feature of the MSE approach is including as much of the known uncertainty as possible.

MSE Concepts Explained Using a Hypothetical Example

- Francis Pantus

The MSE principle relies on calculation of performance measures for each objective to determine how each will be met under certain conditions (e.g. different management strategies). Performance measures provide the means to measure the performance of particular management strategies, but how are they calculated? In order to make these calculations, computer simulations (termed models) are used.

A model is a simplified computer representation of 'reality'. It consists of equations made up of various parameters (in bold in the equation below) representing particular features of the fish stock and the fishery, and the relationship between them. For example

Total Catch = **Biomass** (average weight x number of fish) **x** **Effort** (how many lines/ time spent fishing) **x** **Catchability** (how readily each fish is caught)

Models can be used to simulate and predict whole fishery systems (fishers, the fish, management) over time, and are used to calculate performance measures. Common examples of performance measure include the total catch over a certain period of time, the average catch rate over a period of time, or the extent of depletion of the stock from its unfished state over time.

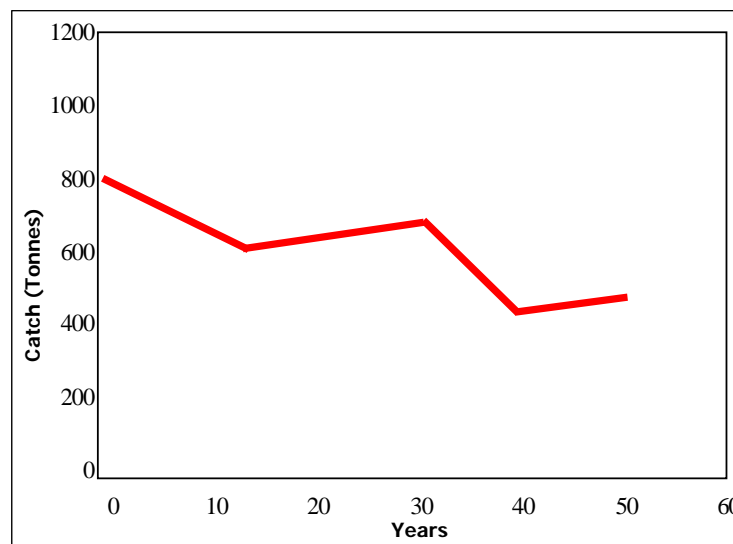
If variability and uncertainty in the fishery being examined are considered to be zero - that is we know everything about the fishery with absolute confidence - every time the calculation is completed the answer will be the same. It would not matter how many times you did the calculations, the result would always be exactly the same.

For the example model given above, if Biomass = 4000t, Effort = 20 days of fishing, and catchability = 0.01% of the biomass per day of fishing, the predicted total catch would always be

$$\begin{array}{rclclcl} \text{Total Catch} & = & \text{Biomass} & \times & \text{Effort} & \times & \text{Catchability} \\ 800 \text{ t} & = & 4000 \text{ t} & \times & 20 & \times & .01 \end{array}$$

If this was calculated for many years the resulting graph would look like that in figure 1. In 50 years the catch would be around 450 tonnes.

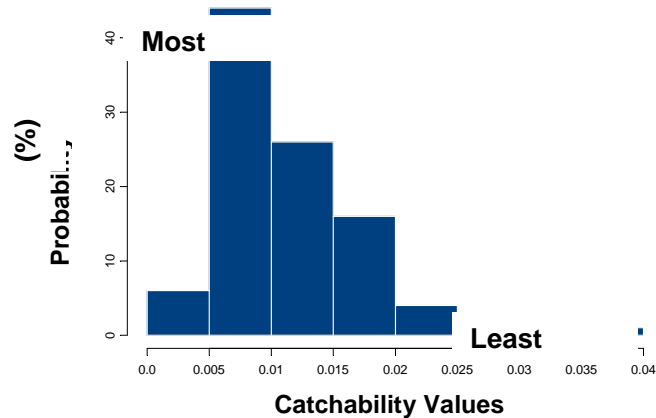
Figure 1: Example of running a model with no uncertainty.



Unfortunately, we know that such a fishery does not exist. In our example fishery described in the equation above, we all know that catchability is not always the same. Many factors affect how easy it is to catch a fish, therefore it is highly likely that catchability will constantly be changing. There are also elements of the fish stock about which we know very little. Therefore, there is a level of uncertainty in the value of the parameters used in such models. For example, there is always considerable uncertainty in the estimated size of fish stocks from one year to the next, similarly the catchability of fish may vary from year to year, fisher to fisher, or one reef to the next.

To incorporate this uncertainty into models these equations are modified to accommodate a range of values for parameters, and also the relative probability that each value in the range represents what is actually happening in the fishery. For example the figure below (figure 2) shows the range (the horizontal axis) of possible values (from 0.0-0.04) for catchability. The height of the bars show the relative probability of each potential value of catchability, with 0.005-0.01 being the most likely value and 0.035-0.04 the least likely. Figures such as these are known as probability distributions and they can be used to describe the uncertainty around particular parameters in models.

Figure 2: Ranges of Catchability Values, and the probability of each of those values



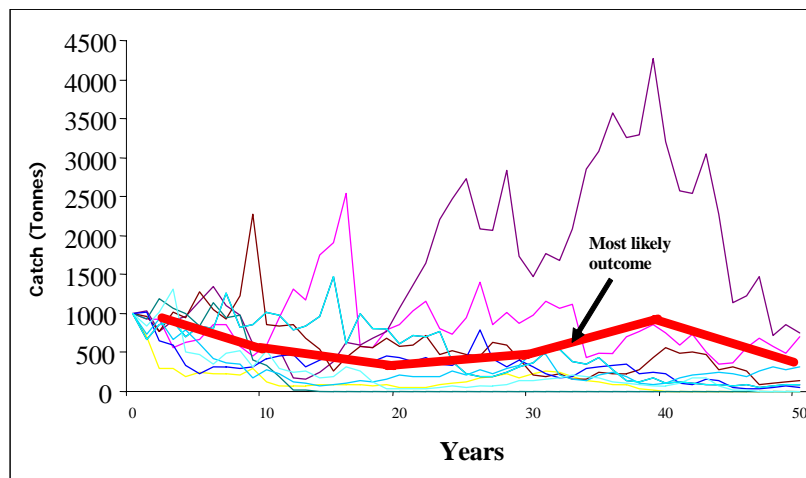
In our example above, if we now consider a range of values for catchability and biomass, and also the probability that each value represents reality in our imaginary fishery, our model will be more complicated:

$$\text{Total Catch} = \frac{(\text{Biomass} \times \text{Uncertainty in biomass}) \times \text{Effort} \times (\text{Catchability} \times \text{Uncertainty in catchability})}{\text{Denominator}}$$

Each time this equation was calculated, a different answer will result, as the values used for each parameter will be a different each time. Models incorporating this kind of uncertainty in the value of parameters are called 'Stochastic Models'.

Running a stochastic model of a fishery 10 times over the same period will give 10 different 'histories' as shown in Figure 3. Running the same model 10000 times gives us a range of values (of catch for the example used here) for each year and their probabilities. The most likely outcome will become evident from calculating the average of all the results (thick line on Figure 3).

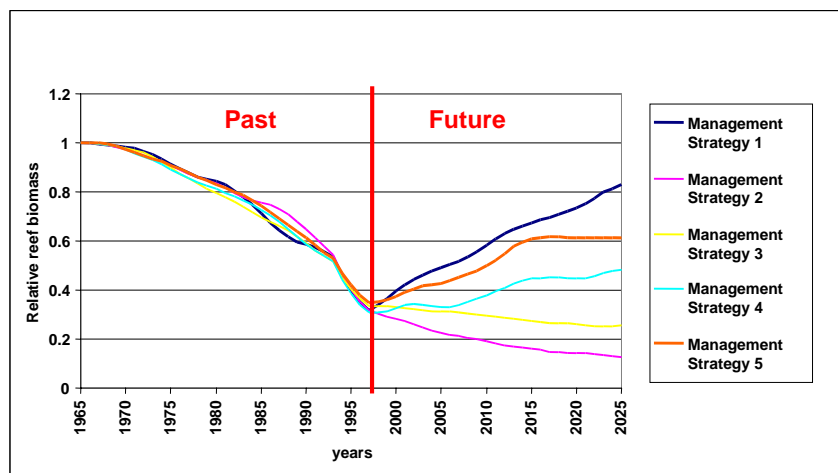
Figure 3: Example of running a model with uncertainty.



Real information (data on catch, effort, biology of target species, etc) from the fishery is used to determine the level of uncertainty, and the relationships between the various components of the model, for example how catchability is affected by the amount of fishing on a reef. The result is a model (set of equations) which as closely as possible can reproduce the real data derived from the fleet and scientific research.

The model is then ready to predict, or simulate, what will happen to the fishery in the future. Figure 4 is an example of future projections for a theoretical fishery under various management strategies using a MSE model. Each line on this graph represents the average result from lots (1000's) of runs of the model, under different conditions, like the thick line on figure 3 represents the most likely result. In this case only the average result has been shown for simplicity.

Figure 4: Example of projections of future conditions of a fishery under various management strategies. Each line represents the average of lots of runs of a simulation.



The outcomes (the average values and the variation around them) from the simulations provides a way to evaluate each strategy and the ability to compare levels of risk for each objective across a number of strategies.

The results can be summarised in a decision table, in variety of ways, and users and advisory committees such as ReefMAC can have input into what ways would be most suitable. An example of a decision table outputted from the hypothetical model is given below.

MANAGEMENT OBJECTIVE

<u>MANAGEMENT STRATEGY</u>	TRENDS IN BIOMASS	MAXIMISE CATCH	AVERAGE CATCH RATES
STRATEGY 1 - EG CLOSE THE FISHERY	STRONGLY INCREASING	N/A	N/A
STRATEGY 2 - EG NO RESTRICTIONS	DECREASING	POOR	POOR
STRATEGY 3 - EG REDUCE EFFORT BY 50%	DECREASING	POOR	MEDIUM
STRATEGY 4 - EG CLOSE 60% OF AREA TO FISHING	INCREASING	GOOD	POOR
STRATEGY 5 - EG REDUCE EFFORT BY 50% AND CLOSE 60% OF AREA TO FISHING	INCREASING	GOOD	GOOD

Issues raised from the floor

It is important to include spatial variability in any models simulating the Queensland Reef Line Fishery.

Response: Such added complexity requires additional information from data collected from the fishery itself and from the fish populations to 'explain' this added complexity in the simulation models. This is a major reason why the ELF Experiment is being done on such a large spatial scale, to obtain the information necessary to encapsulate spatial variation. It is also why we use information from fishers over as broad a scale as possible. These models can also be used to identify key bits of information about the resource system that will reduce the current uncertainty in our knowledge of that system.

The ELF Project and MSE for the Reef Line Fishery

- Campbell Davies

The ELF Project aims to gather basic information on biology of a number of target and non-target reef fish species. Information of interest include; longevity, growth, age at 1st reproduction, age at recruitment. Such information will aid in refinement of existing and proposed fisheries regulations and as input for the MSE models.

The ELF Project is also collating a large amount of information on the behavior of fishing fleets, their responses to changing conditions, and what motivates fishers in their day-to-day decision making. This includes gaining information on fine-scale movements of vessels, economic responses of fishers to changed markets (e.g. live fishing, regional variation in operations) and likely responses to management strategies. This also includes analysis of catch and effort information from the local to regional scales.

The ELF Experiment has a central role in the ELF Project. This experiment will provide important information on the responses of stocks to changes in fishing effort, and the recovery of fish stocks following protection. The ELF Experiment will also provide some important inputs to the MSE models including measures of:

- catchability of target species,
- reef specific biomass,
- relationships between fishing effort and fishing mortality.

Fishing has been an important activity in Queensland for many years, with the commercial fishery being active in the GBR since the 1940's. However there are very few years of comprehensive catch and effort data from the commercial fleet, and only very limited data from the recreational fleet.

What the ELF Project needs from workshop participants is:

1. Specific operational objectives for management of fish stocks and the Fishery.
 2. Identification of measurable performance indicators with which to evaluate these objectives.
 3. Suggestions of alternative management strategies for the fishery.
 4. Identification of major sources of uncertainty within the fishery that we have overlooked.
- Issues raised from the Floor

It was suggested that the information base on recreational fishing effort might be increased from registration data from the Queensland Transport Department, data from Jim Higgs' research on recreational fishing and QFMA's Rfish program.

Response: The ELF team has regular contact with QFMA on estimates of recreational fishing effort. The main issue is getting estimates of spatial distribution of recreational fishing effort on the level of individual reefs, rather than from which port or town. Use of Queensland Department of Transport data is a useful suggestion, though such data need to be interpreted with caution because data such as boat registrations do not necessarily correspond well with levels of recreational fishing in off-shore waters, where the coral reef fin fishery is mostly focused.

User Expectations and Issues in the Medium Term

- Group Discussion

The group sessions were intended to encourage discussion on how general management objectives for the Queensland Reef Line Fishery could be refined into specific operational objectives against which it would be possible to measure performance of different strategies. The participants were then encouraged to suggest strategies that could be used to meet these objectives. The outcomes from these group discussions were then discussed at the workshop level.

A hypothetical example of the desired outcome of this discussion session is given below for the general objective of 'managing the fishery in an ecologically sustainable way'.

Broad Objective: Manage reef fish in an ecologically sustainable way

Operational Objective:	<i>Maintain Common Coral Trout spawning stock biomass (ssb) above a specified threshold level (e.g. 40% of unfished level)</i>
Strategy (1):	<i>Limit effort in fishery to X line-days so that relative biomass does not fall below threshold level</i>
Strategy (2):	<i>Close Y% of the known habitat to fishing.</i>
Strategy (3):	<i>Set minimum legal size so that the estimated SSB protected by the size limit is equal to threshold level for spawning stock biomass.</i>
Performance measures:	<i>(1) Relative abundance of common coral trout SSB (from catch rates or Underwater Visual Survey counts) (For all Strategies)</i> <i>(2) Amount of effort (line-days) in fishery (for Strategy 1)</i> <i>(3) Change in size at 1st reproduction and fecundity by size relationship over time (for Strategy 3).</i>

The objectives and strategies that were discussed in this session are summarised below. Please note that these are objectives suggested by workshop participants for future consideration only, and in no way represent formal endorsement by any particular individual or group participating at the workshop, or by the ELF Project Team.

Objectives

- Maintain healthy and stable catch rates over time and space. For example, catch rates do not decline over three years (by more than 10% per year) in any region.
- Maintain ecosystem diversity
- Have a profitable commercial fishery
- Have satisfied recreational and traditional fishers
- Maintain maximum community benefits from the resource
- No increases in movement or changes in fishing practices by fishers
- Decrease by-catch and or release mortality

- Manage fishery to promote an increase in fish size. It was pointed out that this would be directly at odds with objectives from the commercial fishing sector, and was, therefore, was not a universally agreed objective
- Increase fish biomass
- Maintain fish abundance. For example fish numbers, diversity and size (both for adults and juveniles) increase to levels seen 10-20 years ago depending on region
- Minimise risk of stock falling below a secure reference level - a preserved component of the stock is maintained

Management Strategies

Effort Controls

- Spawning closures
- Limits on the number of active commercial and charter vessels
- Limits on level of recreational fishing effort by use of permits
- Effort reduction of around 50%

Area Closures

- 30% - 50% of reef habitat closed
- Rotational closures of reef habitat focused on 30% of reefs over three years
- Selective rotational closures of 15% of the reef habitat coupled with permanent closures of 25% of the reef habitat and effort reduction strategies for those open areas
- Seasonal strip closures to protect 50% of spawning stock biomass or equivalent area closures with clear open areas and transit lanes for vessels.
- Zoning system to restrict shifts in fishing effort. Would also promote conservation of local area by resident fishers.
- Zoning system with closures within zones.

Limits on catch

- Bag limits on the recreational fishers
- Catch quota for commercial fishers
- Additional size limit restrictions
- Output controls within zones

Other issues raised

- Enhanced enforcement to overcome black marketing of Queensland reef fish.
- Improved equity in regulations.
- Management regulations would benefit by being simpler as it would enhance compliance
- Why use multiple strategies to meet one objective?
- The use of technology to enhance stocks and get them back to desired levels, includes restocking and aquaculture techniques.

A selection of the objectives and strategies raised in this workshop will now be refined and revised to be used as examples for demonstrations of the MSE in future workshops.

MSE for Coral Trout: The ELF Models (ELFSIM)

- André Punt, Campbell Davies, Tony Smith, Francis Pantus

ELFSIM is a computer model designed by the ELF team and developed by team members based at CSIRO Hobart. The model is designed specifically for the Queensland Reef Line Fishery with much of the data used in the model derived from the ELF Experiment, previous research and the fishery. The current software is designed to explore management strategies for coral trout but will be extended to other species in the future. The ELFSIM model is still in the developmental stage. Currently the model works on a spatial resolution of individual reefs and 6'x6' grid squares (approx. 11 km x 11 km) in one month time steps.

There are sub-models within the main model to account for **fish biology**, **dynamics of the fishery**, and **fisheries management**.

The **biological** component of the model takes into account the dynamics of coral trout populations, including growth, mortality (natural and fishing), larval production and movement, recruitment and sex reversal. The ELF Experiment is contributing greatly to the information base for this component of ELFSIM. Each reef is treated as a separate biological population of fish. Adults are assumed to not move between individual reefs, however movement of larval stages between reefs is included in the model allowing the impact of 'sink' and 'source' reefs to be considered.

Including a **fisheries dynamics** component in a MSE model is a fairly novel approach. In the past models have focused on the fish (biology) rather than the fishers themselves. The lack of consideration of the fleet activities has been cited as a common reason for some inaccurate assessments of fish stocks throughout the world. This component of the ELFSIM model will be the most difficult to develop as there are few previous examples from which to work. The basic approach will be to use information from QFMA compulsory commercial logbooks to mimic historic patterns of effort over the regions of the GBR. This will provide information on how effort has been distributed (over time) across the GBR. In the future we will add to this information about fishers motivations and their possible response to changing conditions, such as shifts in markets (live fish trade), changing stock sizes or catch rates, weather (e.g. seasonal variation in trade winds), and new management regulations.

The **fisheries management** component will include information on management constraints that might be placed on fishers. For instance, limits on catch or effort by introduction of any quota system, the extension of closed areas, or introduction of closed seasons. This will allow predictions to be made about how fishers will react to new conditions such as new management strategies, and consequently how these strategies might affect fish stocks.

- Issues raised from the Floor

The issue of the influence of weather on catchability of some fish species was raised with respect to the ability of the model to include this type of variability.

Response: This generally depends on the scale of the weather factor e.g. a south-easterly blow compared with an El Nino event. The consideration of adverse weather halting fishing has already implicitly been including in the model. For example, if certain months have historically been bad for weather, this will be indicated by a decrease in fishing effort and will be reflected in simulations through the fisheries component of the ELFSIM Model.

Closing Comments and Where to From Here

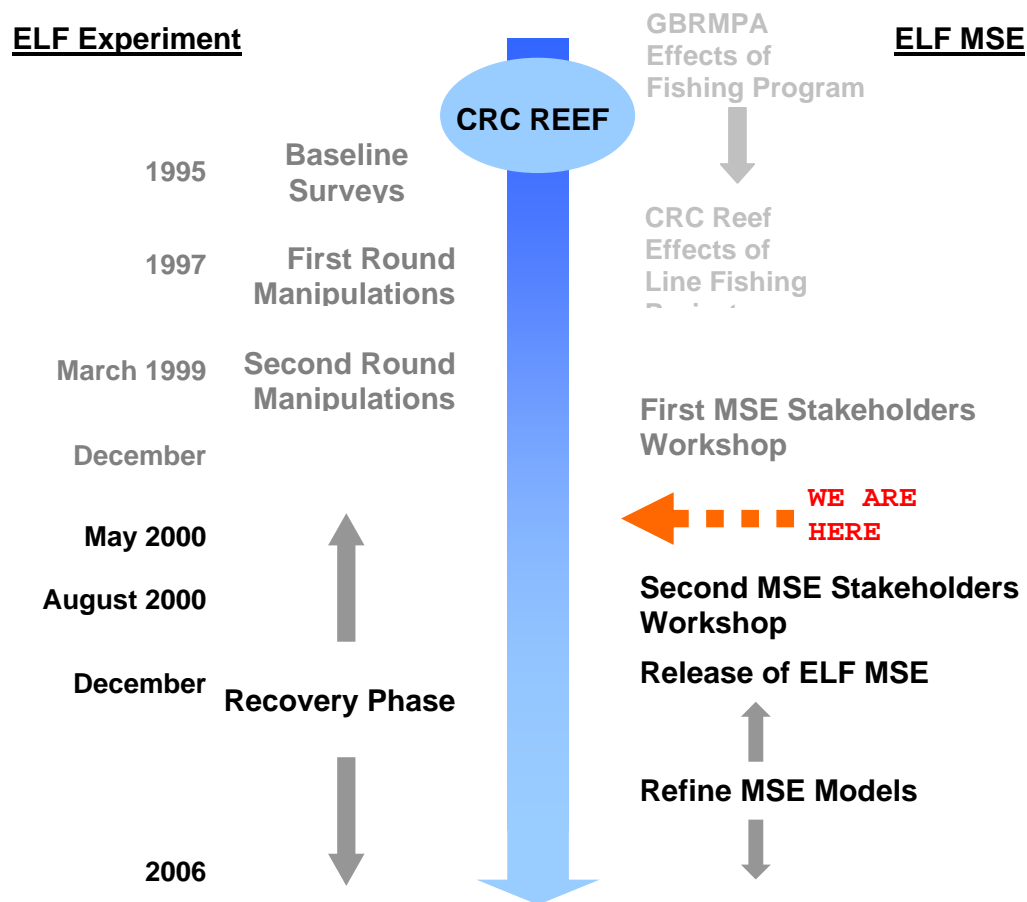
- Bruce Mapstone

The workshop has been a resounding success for the ELF Team. We have gained a tremendous amount from the participation of those who attended, and for this we are very grateful. The suggestions and comments expressed on the day will be of great benefit to the further development and refinement of the MSE models. We will now be able to go away and have a much clearer idea of the types of questions stakeholders will be asking of the models.

The ELF Team will now consider the outcomes of this workshop. We will seek to implement new features, which were raised at this workshop, in the models, for example recruitment pulses, additional areas of uncertainty in the fishery, and alternate management strategies that stakeholders may wish to evaluate using the ELFSIM model.

The outcomes of this meeting will be evident in the next workshop planned for around August, 2000 with the presentation of a more highly developed working model of ELFSIM. The aim of this workshop will be to 'test' the working ELFSIM model and make suggestions for further refinements of objectives, performance measures and management strategies prior to the final launch of ELFSIM in December 2000. We will be contacting participants for further discussions on management objectives before the August workshop.

The ELF Team would like to thank you for your participation in the First MSE Stakeholders Workshop. Your comments and suggestions have contributed greatly to the development of the MSE tools and will be of great benefit to management decision-makers in the future. All participants are welcome to contact the ELF Team if they have any further questions or comments. We look forward to your continued association with this MSE process in the future.



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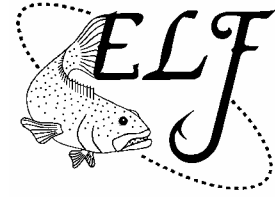
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Appendix E: Report from the ELF MSE 2nd Stakeholder Workshop



SUMMARY REPORT FROM

EFFECTS OF LINE FISHING PROJECT

***MANAGEMENT STRATEGY
EVALUATION***

2ND STAKEHOLDERS WORKSHOP

Southbank Convention Centre

Townsville

14th and 15th November 2000

Report Compiled by

Annabel Jones
Belinda Boyce
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Introduction

The 2nd Stakeholder Workshop on the ELF Management Strategy Evaluations (MSE), held on the 14th and 15th November, 2000, sought to build on the progress made at the 1st workshop and subsequent discussions with participants.

A major aim of this 2nd workshop was to refine and formalise operational objectives and management strategies. Clear objectives and relevant management strategies are essential for the final development of the MSE model, ELFSim. The 2nd workshop also involved discussion of appropriate performance indicators for identified objectives and strategies.

It was also intended that participants would leave the workshop with a clear idea of where the MSE research would go from this workshop to its ultimate release in 2001. New directions for further refinement of ELFSim identified at this second workshop will be explored, as will potential extension activities for reporting the MSE tools to the wider community.

This summary document is intended to provide participants with a record of the workshops proceedings. The ELF team welcomes any comments regarding this document, the 2nd ELF MSE Stakeholders Workshop, the MSE approach, or any other aspect of the ELF Project. Questions or comments can be directed to:

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2nd Workshop Objectives

"To let you know where we are up to and where we are going with ELF tools for Management Strategy Evaluation (MSE), and to get your help to focus the ELF tools on things that matter to you in the reef line fishery."

1. To resolve specific objectives that reflect the needs/desires of stakeholders for management of reef line fishing in the GBRWHA in the medium-long term and which are relevant to the ELF Management Strategy Evaluation tools (ELFSim).
2. To identify potential management strategies expected to realise those objectives in the medium-long term and which are appropriate for evaluation using ELFSim.
3. To specify desirable performance indicators for the above operational objectives and strategies.
4. To further familiarise stakeholders with the MSE approach and features of ELFSim.
5. To clarify follow-up and extension strategies for delivering final results of ELFSim and for future use of ELFSim.

The ELF Project

The ELF Project is based at the CRC for the Great Barrier Reef World Heritage Area (CRC Reef). The research team comprises a collaborative team of researchers from James Cook University (JCU), the Australian Institute of Marine Science (AIMS), the Queensland Department of Primary Industries (QDPI), and the CSIRO Marine Research. In addition, the project has developed strong links with representative industry bodies such as the Queensland Seafood Industries Association (QSIA), Sunfish, and fishers from all sectors of the fishery, and the main management agencies (QFS and GBRMPA).

The ELF Project consists of four core areas of research:

- 5) Fisheries information and fleet characteristics
- 6) Biology of target and non-target species
- 7) Modeling and Management Strategy Evaluation
- 8) Liaison and Information Transfer

Issues for Management of the Reef Line Fishery

The GBR region supports multiple use and is a World Heritage listed area of ecological significance. Management of the region, therefore, must facilitate continued multiple use of the GBR (including fishing), whilst ensuring that such use does not threaten the ecological, aesthetic and cultural values for which it was listed.

The fishery in the GBR Region is accessed by a variety of user groups including recreational fishers, charter boat operators, commercial fishers and non-fishers. These users have access to the same areas, but with some differences in management régimes applied depending on activity. The different sectors have very different motivations and behavior, and consequently different distributions of effort.

The Queensland Reef Line Fishery is a complex one in that it is a multi-species fishery, where a large number of fish species are targeted by fishers in most areas. Coral trout and red throat emperor are the major target species, however, and comprise the majority of the catch in most regions. The biology of many tropical reef species, including the main target species can also complicate management of the fishery. Many species have complex life history strategies (for example many change sex during their life), and their abundance varies from one region of the GBR to another.

Effective management of fishing requires knowledge of the characteristics of the fisheries (catch, effort, fishing practices, economics etc) and of the consequences of fishing for the harvested stocks and the ecosystem. Management of line fishing to date has been hampered, however, by a shortage of information about fishers' activities, their effect on the resource and our knowledge of the biology and dynamics of fish species.

The ELF Project was established to provide the necessary technical base for quantitative evaluations of potential management strategies for line fishing in the GBR environment.

The 1st MSE Stakeholders Workshop

The 1st MSE Stakeholder Workshop initiated essential dialogue among the representative stakeholder groups and developed a firm foundation for refinement of the MSE model (ELFSim) being developed for the Queensland Reef Line Fishery.

This workshop was successful in identifying a number of broad management objectives important to stakeholders and suggested strategies to meet these objectives. In summary these included:

Objectives

- Maintenance of healthy and stable catch rates, both temporally and spatially
- Ensure a profitable commercial fishery
- Maintenance of fish abundance
- Maintenance of a component of fish stocks
- Satisfied recreational and traditional fishers
- Maximisation of community benefit from the resource
- Maintenance of fleet movements and changes in fishing practices

Strategies

- Introduction of spawning closures
- Limit the number of active commercial and charter vessels
- Limit recreational fishing effort by introduction of a permit system
- Reduction of fishing effort by 50%
- Introduction of commercial catch quotas
- Introduction of catch quotas in each zone
- Increase size limit restrictions
- Introduce additional bag limits for recreational fishers

Following this workshop, the ELF team endeavoured to meet with all of the participants to further refine these objectives and strategies in preparation for the 2nd workshop just held.

Meeting Summary

Opening Comments

- Bruce Mapstone

The 1st Stakeholder workshop and subsequent discussions provided much toward to refining ELFSim, setting clear operational objectives and potential management strategies that could meet these objectives. This workshop was therefore an important step in the development of ELF MSE tools which will have a major role in forward thinking management of the GBR fishery in the future. The 2nd Stakeholder workshop now seeks to build on this promising start.

We hope participants will gain from this workshop an awareness of the importance of clear operational management objectives in evaluating management options. In this workshop we also intend to explore the implications of assumptions inherent in the management objectives and in the choice of performance indicators and clarify what uncertainties exist about fish stocks and the exploitation of these stocks.

Clarification of a clear set of operational objectives and management strategies from this workshop will ensure that the outcomes of ELFSim will have maximum relevance and benefit to the Queensland Reef Line Fishery. With these objectives, ELFSim can provide comparative information about stock and catch performance under alternative management strategies. ELFSim will also be able to provide evaluation of how likely strategies are to meet specific objectives.

Thus, the objectives of this workshop are as follows:

- To resolve specific objectives
- To identify potential management strategies
- To specify desirable performance indicators
- To further explore MSE and ELFSim
- To clarify follow up and extension strategies

It was not intended that this workshop would see participating stakeholders reaching consensus on all issues discussed. It is not necessary to have all groups agreeing on strategies or objectives, and this workshop was not the relevant forum to seek such agreement. We do hope, however, that it will provide a vehicle for free constructive deliberation of a wide range of options for management of the reef line fishery. Also, this workshop was not the arena to discuss the Queensland Fisheries Services Draft Management Plan for the Coral Reef Fin Fishery, although ELFSim will be a valuable tool for consideration of reviews of such plans in the future.

Finally this workshop is a starting point, rather than the final word as far as ELFSim or management objectives for Reef Line Fishery are concerned. ELFSim and the ELF team can provide assistance in this process by developing and applying the MSE approach, but the process will be an ongoing one that depends substantially on the participation of stakeholders.

Refresher Talk on ELFSim – Worked Example

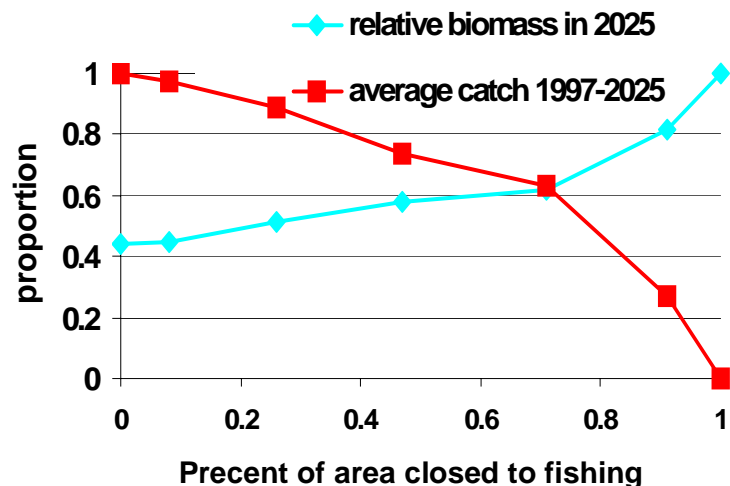
- Tony Smith

A worked example using ELFSim was presented to the stakeholders to remind them of the MSE process and its strengths and weaknesses. This example was designed to demonstrate the capability of ELFSim and the MSE process to predict consequences of future management strategies for coral trout. While ELFSim cannot exactly simulate the fishery or how stocks of coral trout will respond to potential changes to the environment, it can predict the most likely outcome.

An area of the GBR in the Cairns region was considered in the example. The (hypothetically) current status of the reefs in this area vary with respect to fish stocks, ranging from close to untouched (virgin stocks) through to reefs that are quite heavily exploited. The hypothetical management strategies considered in this example considered closure of a range of reef areas, from all closed to all open (0, 8, 26, 47, 71 or 100% of the area). ELFSim estimated the effect of each management strategy on the fishable (exploitable) biomass over the next 25 years (see figure 1).

Interestingly, the projection indicated that catch did not drop at the same rate as the number of reefs open to fishing (Fig. 1). This is most likely due to the fact that as reefs close fishing effort shifts to those areas that remain open. Conversely, total biomass did not increase at the same rate as the increase in number of reefs closed. It is important to note that currently ELFSim does not have a feedback loop relating total fleet effort and changes in fishing behaviour to area closures. Hence, effort remains at a pre-specified level, even if only a few reefs are left open to fishing. This means that as more reefs are closed to fishing, all the effort is increasingly crammed into the diminishing number of reefs left open.

Figure 1: Projected average catch and relative biomass of a region of reefs of Cairns with various amounts closed to fishing. Note the average biomass does not increase at the same rate as area is closed to fishing. This is due to fishing effort shifting to open areas as reefs are



In the absence of any management objectives against which to measure each option, it was unclear as to which option (amount of area closed) was most appropriate in the example presented. This clearly demonstrated the need for operational objectives, with clearly specified performance measures against which the potential success or failure of each strategy can be evaluated. In the absence of operational objectives, ELFSim will only be able to provide stakeholders with an assortment of summary statistics such as catch, catch rates etc. This abundance of additional data alone will not necessarily make the process of evaluating different and possibly conflicting options easier.

If reference levels, or desirable targets, can be agreed for performance indicators, however, ELFSim can estimate the probability of exceeding those reference levels under different management strategies. This will allow direct comparisons of the relative performance of different strategies against a range of competing objectives. If the desired probability of exceeding the reference levels can also be pre-specified, the results can be expressed simply in terms of whether the objective was satisfied or not by each strategy. [This step is probably beyond our collective capability at this stage for the reef line fishery.]

The results from each hypothetical evaluation were presented as a decision table. This table lays bare the trade-offs among different strategies in the same 'currency', allowing comparison of the performance of each management strategy relative to each other strategy. The key advantage of a decision table is that it provides a clear synthesis of a great deal of information, providing a framework for useful consideration of the different management options.

In conclusion, when setting objectives, important considerations are:

- Which variables are to be used (eg spawning biomass versus fishable biomass)
- The period of time to consider for evaluations (eg the situation at the end of the period or the average over time).
- Choice of target level (eg 50% of virgin stock for an indicator based on spawning biomass).
- The probability that the performance indicator exceeds the target level.

Objectives and Performance Indicators from Stakeholders

- Campbell Davies

In discussing operational objectives with participating stakeholder groups prior to this workshop participants generally erred on the side of conservatism and had some difficulty "putting numbers" on objectives. However, participants felt that operational objectives should be regional as well as GBR wide and should reflect the general desire to maintain stocks in a relatively "natural" state. The *objectives identified by stakeholders prior to the workshop* mostly fitted into three common categories.

1. Objectives for Coral Trout Stocks

- Some reasonable proportion of stock remains protected from the fishery
- Maintain sex ratios
- Ensure some proportion of the stock was allowed to reach maturity and spawn each year

- Ensure adequate representation of all ages/sizes in population
 - Minimise disruption to spawning
2. Catches of Coral Trout (All catch objectives were contingent on stock objectives outlined in 1. above)
- Allow maximum catch to be taken
 - Ensure no significant reduction in 1:1 ratio of legal:undersize coral trout caught over any three year period
 - Ensure sizes of fish in catch satisfy stakeholders' needs (a possible surrogate for social objectives which can not be assessed by ELFSim directly)
 - 100% probability of catching a coral trout > 2kg in all/any years
3. Stability of Catches of Coral Trout
- Minimise risk of decline in potential yield in long term (Decades)
 - Minimise disruption to the harvest
 - Range of inter-annual variation should not exceed 10% of long term average

It was noted that fish stocks (biomass) will not return to virgin condition in the short term even if an area is closed. Therefore recovery-related performance measures should be considered over the medium to long term. However, for catch, it is necessary to consider short-term as well as medium and long-term effects.

Several of these objectives were discussed in further detail to determine their importance, accuracy in reflecting stakeholders requirements, and to aid refinement where required.

Objectives for Coral Trout Stocks

OBJECTIVE 1

Biomass of coral trout in *closed area*

Broad objective: Maintain some proportion of coral trout stocks in closed areas at an unfished state in all regions of GBR.

Operational Objective: Ensure that the mature biomass in closed areas does not fall below 'X' % of the unfished mature biomass in any region of GBR.

This needs to be refined further before evaluation by ELFSim ie a value for 'X' is required. Campbell suggested 'X' = 70% or 80% to initiate discussion.

Also, two secondary objectives came to light during discussions.

Secondary objective 1: Ensure closed areas are effective in protecting coral trout stocks from fishing, ie the catch from the closed areas should be zero (this is a performance measure (Pi) for secondary objective)

Secondary objective 2: Maintain sex ratio of stocks in closed areas to within limits of unfished stocks (regionally).

It was generally agreed that if stock biomass was maintained at adequate levels sex ratio would be maintained also.

A suggestion for 'X' of 80% to 90% was made by participants. Given fishing is allowed in open areas of the GBR and some level of infringement at the margins of open/closed areas will occur, the biomass would not be expected to always recover to 100% of unfished levels in closed area.

A value of 'X' = 80% was suggested as a bottom-line (threshold level) which, if transgressed, would precipitate immediate action. Participants felt that the biomass in closed areas should be above this level at least 90% of the time. A further suggestion set a target level rather than a threshold level of 'X' = 90% around which biomass could fluctuate.

Action: The ELF team will evaluate the operational objective and report on the average mature biomass relative to the unfished level as well as the probability that the mature biomass in the closed areas exceeds 80% of the unfished level over the last 5-10 years of simulation.

OBJECTIVE 2

Biomass of coral trout in open areas

Broad Objective: Maintain sustainable stocks of coral trout in open areas in all regions of GBR.

Operational Objective: Ensure that the mature biomass of coral trout in open areas does not fall below 'Y'% of the unfished mature biomass in any region.

Again this operational objective requires refinement in selecting a value for 'Y'. Campbell suggested values of 'Y' = 20%, 40% or 60%. There was some discussion around these values, resulting in the conclusion that none of us had sufficient knowledge at this stage to stipulate what would constitute a single most sensible value. Thus, it was agreed that ELFSim would be used to provide feedback on performance of different strategies against a range of potential reference levels (as indicated below).

Action: It was agreed that the probability of the mature biomass being above 30%, 40%, 50% and 70% of the unfished level would be reported for simulations, together with the actual values of biomass.

Secondary objective: Ensure 'adequate' representation of all ages and sizes of fish in the open areas, compared to those present in the unfished stock. Performance Indicator -average size/age of fish relative to unfished stock or the sex ratio of mature population relative to unfished stock.

The ELF team indicated that sex ratios are naturally highly variable and may not be a sensitive indicator of performance against objectives. It was also pointed out that sex ratios were likely to be highly susceptible to fishers selectively targeting a particular size of coral trout (and therefore more of one sex than the other). For instance, the live fish trade prefers plate size coral trout. Such size-specific targeting would almost certainly lead to misinterpretation of sex ratio as a Performance Indicator. ELFSim can incorporate targeting of particular size of fish as well as gear selectivity to evaluate such effects, although current information on gear selection is limited to the gear most used in the commercial fishery. ELFSim will need more information on other gear type (eg hook sizes) to incorporate gear selectivity over a whole range of gears.

It was suggested, therefore, that the average age of catch would be a more sensitive measure of change in population structure than the mean length of the catch or the sex ratio.

It was noted, however, that measuring the mean age of the catch in the field would be difficult, but that the PI could still be used usefully as part of ELFSim evaluations.

A range for 'Y' (= ratio of average or size in catch to average age or size in unfished population) from 35% to 70% was suggested by participants.

OBJECTIVE 3

Catch of coral trout from open areas

Broad Objective 1: Maximise catch of coral trout (given stock objectives above)

PI = average catch of legal size of coral trout/year in open areas.

Broad Objective 2: Minimise variability in catches of coral trout from open areas.

PI = Average of (catch this year – catch last year)/Average annual catch

These objectives were not refined to any finer detail.

Catch rates of coral trout from open areas

Broad Objective: Maintain **minimum** average catch rates in open areas above economic/satisfaction threshold of (initial values for discussion):

For commercial fishers 15 coral trout/dory/day

For charter fishers 2-4 coral trout/client/day

For Recreational fishes 2-4 coral trout/fisher/day

These target levels are a suggestion only, and there was much discussion of how appropriate they were. In selecting values, it is necessary to distinguish between thresholds/bottom lines (levels below which a commercial fisher would no longer consider the fishery viable, or a recreational fisher would no longer go fishing) and targets. The PI for this objective was suggested to be average catch rate.

This PI could be considered a surrogate for the economic viability/satisfaction aspects of the fishery that can not be evaluated directly by ELFSim.

Discussion

The reference level set for the commercial fishers will probably most appropriately be a bottom line, as commercial success has a clear bottom line beyond which fishers cease to make a profit. For charter and recreational fishers, however, a reference level will probably be more appropriately set as a target level that would indicate satisfactory catch rates.

It was suggested that a target level for commercial catch rates that would be acceptable would be around 20–30kgs coral trout/dory/day (around 20-30 fish). A further suggestion was that target reference levels for the commercial fishery could be set from historic catch records during periods that were generally considered to be profitable. For instance, catch rates ('Y') for evaluation of this objective could be set at 80% of average commercial catch rates over the last three years of fishing.

It was pointed out that there is a need to consider this figure over all commercial operations, ie large and small operations and those supplying the markets for live and dead fish. The ELF Team was questioned by stakeholders on whether ELFSim could report on different components of the commercial fleet to capture the impacts of management strategies on both dedicated and occasional line fishing operations (eg trawlers, net fishers etc who only

line fish opportunistically). This would be possible in the model, but requires some modification to the ELFSim software. Given that management strategies do not consider managing these components of the commercial fishery differently, it was noted that this additional information might not provide much to the management debate at present. Nevertheless, the ELF team will investigate this modification.

Regional issues were important for both the charter and recreational fishers. An example was presented to the meeting of charter fishers from the Cairns and Swains region. In Cairns catches of 1-2 coral trout/client/day were considered acceptable. However, in the Swains region, catches of 4-6 coral trout/client/day would be acceptable. To capture this range of values for acceptable catch rates from various regions of the GBR, it was suggested that the ELF team report on performance of management strategies against reference levels ('Y') of 1 and 5 coral trout/fisher/day for charter and recreational catch rates.

Action: For commercial catch rates, target reference levels ('Y') will be set at 80% of average catch from 1993-96. For charter and recreational sectors, target reference levels ('Y') will be set at 1 and 5 coral trout/fisher/day. The ELF team will also report back on the possibility of ELFSim being able to incorporate different components of commercial operations.

Action: Other performance indicators the ELF team will report back on include:

- Catch for all fishing sectors in open areas
- Average age of the catch compared to average age of the catch in an unfished state
- Sex ratio relative to the unfished sex ratio. This may not be very informative as there appears to be quite a bit of variability in sex structures throughout the GBR, and even within regions. We need more information on sex structures of coral trout before this measure may be more informative.
- Average year to year variation in catch
- Relative proportion of big fish (>45cm FL) in the catch.

There were also some objectives proposed by stakeholders that ELFSim will not be able to evaluate. These were:

- Biological features eg maintenance of genetic diversity
- Social Factors eg ensure equitable access to fish stocks and enhance economic benefits of recreational fishing and tourism
- Regulation such as introduction of capped charter vessel licences.

It was emphasised by several stakeholders that although the MSE process can't consider them, such objectives are important points to be considered by advisory groups such as ReefMAC in consultative processes. Highlighting these objectives can emphasise areas of the fishery that may require further research.

General Discussion of Objectives

In discussion of the above objectives it was clear that stakeholders need to prioritise the objectives which they feel are important to the fishery. This could provide a hierarchical framework, which could streamline the process of selecting objectives for the MSE process.

Participants should consider if they wish PI's to be set as threshold or target levels. A threshold level should be considered the bottom line, beyond which the fishery or stocks would be considered in an unacceptable state. It generally would be considered extremely undesirable that a PI would fall below this level, and hence the risk of falling below a threshold level should be very low. However, if target levels are chosen as PI's, there should

be a greater tolerance of falling below the reference level, that is, there should be acceptance of a greater risk of falling below this level.

The values listed above are only suggestions, and hence a starting point for further analysis and discussion. There was concern that any target levels agreed in this forum would become the 'benchmark' for management. This workshop is not intended to be a management forum and ELF scientists reiterated that details discussed here are a starting point only. Following evaluation by ELFSim and reporting back to stakeholders, the objectives and performance indicators will be refined. The MSE process aims to assist managers and advisory groups to set appropriate targets, by providing information, not the answer. How stakeholders respond to results from the MSE process is a management planning issue that should not impinge on the use of the research to inform future decisions.

ELFSim 'on the fly' – What it can do

- Andre Punt/Rich Little

ELFSIM is a computer model designed by the ELF team. It is designed specifically for the Queensland Reef Line Fishery with the data used in the model derived from the ELF Experiment, previous research and the fishery. The current version of ELFSim is designed to explore management strategies for coral trout but will be extended to other species in the future. ELFSim is still in the developmental stage. Currently the model works on a spatial resolution of individual reefs and 6'x6' grid squares (approx. 11 km x 11 km) in one-month time steps.

There are sub-models within the main model to account for fish biology, dynamics of the fishery, and fisheries management.

The front end of ELFSim was run through with participants to provide a tangible demonstration of what can be included in ELFSim, and how simulations are set up. The parameters that determine the properties of each simulation can be set via a series of input screens covering the following subjects.

Runtime Parameters

These parameters specify the time period to consider in the model. The model includes a method for introducing external features that can effect natural mortality of coral trout (positive and negative), including climatic effects eg global warming, and other factors such as recruitment pulses.

Population Parameters

These parameters allow the user to specify the extent of variation in natural mortality of coral trout between reefs and at different times. They also allow tuning of the initial state of the resource and the level of depletion at some period after historic fishing (eg in 1996).

Spatial Structure.

ELFSim models the coral trout resource on the GBR to the level of individual reef. Areas of the GBR can be selected, and within these areas, reefs can be selected to be closed or open to fishing. Once an area is selected, ELFSim can output historical catch for that area. It should be noted actual catch data is available from 1986 through to present, and that it is necessary therefore to project catches back to 1965 in a linear fashion. 1965 is the year in which the line fishery was assumed to have started (for ELFSim purposes). This starting date could be changed, but there would be little advantage to making it earlier and such a change would increase the time taken to run the simulations.

Effort Allocation and Fleet Effort

This screen is where the effort model of ELFSim is represented in the user interface. Future levels of effort (for the whole GBR and/or by region) for the commercial, charter and recreational fleets can be specified here, as can the relative catchability coefficients for the three sectors of the fishery.

Output

Any reef can be selected and ELFSim will output a number of graphs reporting various PI's over all years of history (to 1965) and future projection (to whatever year is set as the end point of future scenarios). Alternatively a particular year of the projection can be selected and the spatial variation in a PI graphed. One suggestion from the previous stakeholder workshop was that a measure of the proportion of big coral trout (>45cm fork length) in the catch was important. This measure is now implemented in ELFSim and can be reported for the charter and commercial fleet.

In summary, there are many options available in ELFSim allowing a very large number of scenarios to be considered. The negative aspect of this flexibility, however, is that without clear operational objectives, and prioritising strategies to evaluate, the evaluations will be either unfeasible (due to the required computing time) or uninterpretable (due to the complexity of the final decision tables). This again highlights the need to clearly set operational objectives and strategies that stakeholders wish to evaluate with ELFSim.

Management Strategies favoured by Stakeholders

- Bruce Mapstone

The range of strategies that had been tabled at the first MSE stakeholder workshop was reviewed and, together with subsequent discussions with stakeholders, were grouped into four main categories:

- Area Closures
- Effort Controls
- Size Limits (minimum and maximum)
- Catch Limits

Strategies which can not be evaluated by ELFSim at the moment

Some of the desired strategies were not amenable to evaluate by ELFSim at this stage:

- Bag limits and regional or area specific bag limits
- Commercial catch quotas (except for olympic type quotas). Catch quotas were not considered a priority strategy as they are not currently being considered for the Queensland Reef Line Fishery.
- Minimum gear restrictions (eg hook size).
- Individual day quotas for the commercial fleet

Discussion

Gear restrictions could be incorporated in ELFSim. However we do not have data at present on the selectivity of possible alternative gear types.

A surrogate was suggested to test bag limits. Calculation of the distribution of catch of recreational fishers could be used to determine the effect of a lower bag limit by removing only the effort of those fishers which catch over a new bag limit. To model bag limits directly you really need information on individual behaviour of fishers and an individual based model. Neither is currently available. Similarly, individual day quotas for the commercial fleet cannot be evaluated, as it requires information on individual operations that is not available in ELFSim currently.

Things that we can evaluate by ELFSim

The following broad classes of strategies can be assessed by ELFSim now:

- Area closures. (Likely to be a primary strategy of GBRMPA for some time to come with GBRMP and RAP). Suggested at 30% -50% of GBR reefs closed to fishing as a starting point for discussion of potential strategies.
- Rotational area closures (eg 30% of GBR closed to fishing at any one time on a three year rotation; 15% at a time on ? year rotation + 25% reef closed permanently).
- Seasonal (Spawning) closure (eg Large strips closed over spawning season; Complete closure for part/all of spawning season).
- Size limits (minimum and maximum)
- Effort limits for commercial and charter sectors, both over whole GBR and also regionally

ELFSim currently assumes that effort not used during spawning (or other seasonal) closures will be re-allocated at other times of year, and therefore total effort is unchanged by these strategies. If stakeholders also want to reduce effort this must be made explicit in strategies.

Final suggestions for strategies to be evaluated by ELFSim

A range of 9 alternative strategies (or combinations of strategies) that captured a range of stakeholder suggestions was tabled to stimulate discussion. After some discussion, the following set of strategies was agreed to form the basis of the MSEs that would be done to complete this stage of the ELF Project.

1: Status Quo – Current levels of effort, area closures and current size limits

Area Closures

2: No area closures, with current levels of effort and current size limits

3: Area closures increased to 50%/region with current levels of effort and current size limits
This strategy represents the RAP strategy

Effort Limits

4: Increase effort to 1.5 x 1996 level of effort with current area closures and size limits

5: Reduce effort to 1996 level of effort with current area closures and size limits

6: Reduce effort to 50% of 1996 level of effort with current area closures and size limits

Seasonal Closures

7: Spawning season closures with effort at 1996 level and current area closures and size limits

This strategy evaluates strategy proposed in QFS Draft Management Plan

Size Limits

8: Size limits changed to 50% size of maturity with current levels of effort and area closures

9: Size limits changed to 50% size of maturity with 1996 effort

Examining the unknowns and testing key assumptions

- Tony Smith

There are at least two types of uncertainty

- Random events such as variability in recruitment and natural mortality. The implications of these can be assessed, but this type of uncertainty cannot be reduced by research.
- Uncertainty in key model assumptions such as relative depletion of the resource after a historic period of fishing (eg to 1996). These can be reduced with more research information.

Sensitivity tests provide information on the effect of the values we chose for sources of uncertainty and model assumptions. Some will make a difference to the performance measures, some will not.

Based on previous experience, the key biological uncertainties which the ELF team will investigate further include:

- Current levels of relative biomass
- Resilience of populations (ability to recover from depletion quickly).

For each of the strategies evaluated by ELFSim, four scenarios (two alternative values for each of the key biological uncertainties) will be run to examine the effects of the above uncertainties on the conclusions we draw from ELFSim.

The question was raised as to whether ELFSim could also look at sensitivity to 'management uncertainty'. For example, what effect infringements could have on the ability of a particular management strategy to meet a set objective. Hence it would be useful to evaluate the sensitivity of various strategies to varying levels of infringement. Participants suggested that levels of infringement that may be meaningful would be 5% and 10%, that is in a closed area fishing effort would be set at 5% or 10% of the fishing effort recorded in an adjacent area that is open to fishing.

Where to now?

- Bruce Mapstone

This Stakeholder Workshop was been highly successful in identifying clear operational objectives and management strategies to meet these objectives. The ELF team will now undertake to evaluate these scenarios with ELFSim. Input from participants was sought as to how to report the results of these evaluations back to stakeholders.

Suggestions included:

- 3rd Stakeholder Workshop planned for early 2001
- Formal Reports to funding bodies including CRC, FRDC and GBRMPA
- Summary report from this workshop
- Articles in the popular press and ELF Newsletter
- Oral presentations to stakeholder groups on request
- Scientific publications
- Information brochure

It was emphasised by stakeholders that careful consideration must be given to how information resulting from the MSE work is to be distributed. In particular, some of the scenarios that were discussed at this workshop are to be explored to capture a wide range of responses to potential management scenarios and it must be emphasised that these particular scenarios were not being recommended as the favoured strategies for managing the fishery. Despite the concerns, however, it was generally agreed that it was more important to distribute the information in an appropriate way rather than to withhold the information. It was suggested that the ELF Steering Committee may be a useful vehicle for ensuring distributed information is appropriate and for managing potential misunderstandings of the MSE results.

There was discussion about future requests for evaluation of management strategies by ELFSim. ELFSim is a complex computer model and although it will be refined in future to be more streamlined its use will continue to require expertise and have high computational capacity. Hence it was agreed that a formal process for evaluating requests is critical to streamline the process. ReefMAC and/or the ELF Steering Committee were suggested as appropriate organisations through which to channel requests for future evaluations. The ELF Team will devise a formal process and report back to stakeholders at the next workshop.

Closing Comments

- John Kerin closed the workshop.

Participants

Independent Chair of QFMA and ReefMAC	John Kerin
Commercial Fishing	Cliff Greenhalgh Bill Weekes
Conservation Interests	Carol Booth Jeremy Tager Anne Ferguson
CRC Reef Research Centre/FRDC	Russell Reichelt
FRDC	Alex Wells
GBRMPA	Mick Bishop
QFMA	Rosemary Lea Mark Elmer Danny Brooks
QSIA	Duncan Souter
Recreational Fishing	Vern Veitch John Robinson
Research	Bruce Mapstone Campbell Davies Annabel Jones Tony Smith Andre Punt Francis Pantus David McDonald Rich Little David Williams Bridget Green Mikaela Bergenius Belinda Boyce
Administrative Support	
Apologies	
Charter Fishing	Bill Edwards Bruce Stobo John Evetts Peter Todd
Commercial Fishing	Robin Stewart
Conservation Interests	Eddie Hegerl
Day-to-Day Management	Bob Grimley Phil Cadwalleder
GBRMPA	
Recreational Fishing	John Tanzer
Seafood Processor	Graeme Vallance Graham Caracciolo

Glossary

Biomass	Measure of weight of fish in a stock
Mature Biomass	Biomass of those fish which are reproductively mature
Fishable Biomass	Biomass of fish vulnerable to fishing gear eg hooks
Spawning Stock Biomass	Measure of total weight of all reproductive fish in a stock
Relative Biomass	Biomass of stock in a particular year relative to biomass before fishing started (B/Bo)
Operational Objective	
Management Scenario	
Management Strategy	
Performance Measure	A status target summarizes how well a specific management objective has been achieved
Decision Table	Shows how well a range of strategies achieve management objectives
Risk	Chance (probability) of an event
Robust management strategies	How well strategy works under uncertainty

Appendix F: Intellectual Property

No patentable or marketable products or processes have arisen from this research. All results will be published in scientific and non-technical literature. The raw data from compulsory fishing logbooks and voluntary fishing diaries remains the intellectual property of the Queensland Fisheries Service. Raw catch data provided by individual fishers to project staff remains the intellectual property of the fishers. Intellectual property accruing from the analysis and interpretation of raw data, computer models and simulation results vests jointly with the Fisheries Research and Development Corporation, The CRC Reef, CSIRO James Cook University, the Great Barrier Reef Marine Park Authority and the Principal and Co-Investigators.

Appendix G: Staff

Principal Investigator:	Bruce Mapstone
Co-Investigators:	Campbell Davies, Annabel Jones Rich Little, Dong Chung Lou David MacDonald Anthony D.M. Smith, Francis Pantus, Andre Punt, Garry Russ, Ashley Williams
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