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The biology, ecology, and management of corals targeted for commercial harvest by Australian coral fisheries

Thesis submitted by

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Statement of the contribution of others

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Chapter	Contributor	Nature of Assistance
Chapter 1		
	Prof. Morgan Pratchett[1]	Manuscript writing and editorial assistance
Chapter 2		
	Prof. Morgan Pratchett[1]	Attaining data, conceptualising analysis, conceptual/manuscript writing and editorial assistance
Chapter 3		
	Prof. Morgan Pratchett[1]	Data collection, conceptualising analysis, conceptual/manuscript writing and editorial assistance
	Dr. Ciemon Caballes[2,1]	Data collection, conceptualising analysis, conceptual/manuscript writing and editorial assistance
Chapter 4		
	Prof. Morgan Pratchett[1]	Data collection, conceptualising analysis, conceptual/manuscript writing and editorial assistance
	Dr. Ciemon Caballes[2,1]	Data collection, conceptualising analysis, conceptual/manuscript writing and editorial assistance
Chapter 5		
	Prof. Morgan Pratchett[1]	Data collection, conceptualising analysis, conceptual/manuscript writing and editorial assistance
	Dr. Renial Cabral[1]	Conceptualising analysis, conceptual/manuscript writing and editorial assistance
	Dr. Ciemon Caballes[2,1]	Data collection, Conceptualising manuscript
Chapter 6		
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Abstract

Coral reefs are highly diverse, abundant, and productive ecosystems, and are a major source of exploitable natural resources. However, tropical coral reef ecosystems, and extractive industries that rely on these systems, face unprecedented environmental and social pressure due to sustained and ongoing degradation. Increasing environmental and habitat changes conflate the perennial and pervasive threat posed by fisheries exploitation. Globally, large scale industrial fishing is recognised as one of the foremost threats to biodiversity and ecosystem function. However, the significant impacts, such as local depletion and extirpation, posed by small scale fisheries is also becoming increasingly apparent.

Marine aquarium fisheries are small scale and highly selective, relaying on hand collection of individual organisms. However, the cumulative scale of coral harvest fisheries is significant. Hard corals (order Scleractinia) are one of the foremost taxa harvested for sale as marine aquarium organisms. Ornamental coral fisheries not only have significant potential to affect the abundance, demography, and phenotypic diversity of targeted coral species, but can directly affect the structure and function of coral reef ecosystems. Sustainably managing the potential impacts of coral harvest is challenging, but increasingly important given the escalating threats facing reef ecosystems and widespread habitat degradation.

Australia has historically been a minor contributor to the global aquarium coral trade, however harvest levels have increased markedly in recent years, especially since 2018. Australia’s fisheries have long been regarded as well managed and presenting limited threat to wild coral stocks, mainly owing to the seemingly small harvest levels and limits compared to the presumed abundance of targeted coral taxa and the large geographical extent of Australia’s coral reefs. However, due to the difficulty of linking commercial catch with environmental impact, Australia does not currently have quantitative, ecologically informed

harvest limits. Furthermore, increasing anthropogenic pressures and disturbances (especially, climate change) may have undermined the sustainability of Australian coral fisheries, but certainly increased environmental, public, and political scrutiny of fisheries management.

All hard corals are listed in appendix II of the *Convention on International Trade in Endangered Species of Wild Fauna and Flora* (CITES), such that international trade is contingent on the provision of sufficient evidence to demonstrate negligible effects of harvesting on targeted taxa and ecosystems. To meet the requirements of this process, state managed Australian coral harvest fisheries undergo periodic reassessment and for Wildlife Trade Operation (WTO) approval of export (i.e., federal/internal) and import (i.e., international/external). Despite ongoing export approvals, there is increasing international concern regarding the sustainability of coral harvesting in Australian water, which has resulted in import bans of Australian (in particular, Queensland) corals.

The Queensland Coral Fishery (QCF), Australia's largest coral harvest fishery, has experienced unprecedented and seemingly unmitigated growth in harvest. Harvest levels of hard coral (Order Scleractinia) increased from just over 56,000 pieces/21,000 kg in 2006/07 to over 490,000 pieces/93,000 kg between in 2019/20. This has been driven by escalating harvest levels of several key species. For example, harvest levels of *Trachyphyllia geoffroyi* increased from just under 3,000 pieces/ 168 $kg \cdot yr^{-1}$ in 2006/07 to over 29,000 pieces/1,264.8 $kg \cdot yr^{-1}$ in 2019/2020. The overall annual quota for the QCF is 200 tonnes, which is further split into 70% 'other corals' (i.e., small polyp stony; SPS) and 30% 'specialty corals' (i.e., large polyp stony; LPS). While overall harvest levels are still within quota limits, annual harvest levels have increased 750% for retained number of coral pieces and 186% for total retained weight from 2006-07 to 2019-20. While more recent analysis has been conducted to determine the current state of coral harvest fisheries, due to the high number of taxa within these fisheries, further focus is required to examine trends relevant to those taxa most at risk due to either exploitation, biological/ecological vulnerability, or both.

Acropora corals are the dominant group of hard corals harvested in the QCF, accounting for over 30% of total harvest in 2019-2020. Though highly abundant throughout the GBR, these corals are also extremely vulnerable to fisheries-independent threats, especially climate-induced coral bleaching. Attempts to understand the effects of coral harvesting is complicated as harvest limits are based on weight, while there is very limited data on standing biomass of *Acropora* spp. Therefore, I established size-weight relationships for a total of 3,972 colonies across six different growth forms (bottlebrush, caespitose, corymbose, digitate, staghorn and table) composed of whole and fragmented colonies. These relationships were then used to estimate harvestable biomass of *Acropora* spp. and better understand the ecological context of current weight-based harvest levels and limits. Using these relationships, and field-based sampling at 12 sites across seven distinct reefs, *Acropora* spp. biomass of ranged from an estimated 0.12 kg.m^{-2} to 4.7 kg.m^{-2}. These estimates necessitate further consideration of catch composition and the specific abundance of individual species that are heavily harvested, and how impacts of current harvest practices interact with species-specific vulnerability to climate change and other escalating human pressures contributing to the degradation of coral reef ecosystems.

Reported harvest levels for individual genera/species of LPS corals are small compared to *Acropora*, but it was among these taxa that the most significant increases in harvest levels have occurred. LPS species have also been considered to face increased risk of local depletion (when compared to branching taxa such as *Acropora*) due to their slower growth. Some heavily targeted LPS taxa harvested in Australia are also single polyp (i.e., monostomatous) species, meaning that taxa must be harvested as individuals only. To develop capacity to monitor harvest impacts *in situ* for these particularly vulnerable species, size-weight relationships were modelled based on extensive sampling of corals (2548 individuals/colonies) for some of Australia's most heavily targeted coral species (*Catalaphyllia jardinei*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Homophyllia* cf. *australis*, *Micromussa lordhowensis*, *Trachyphyllia geoffroyi*). These data facilitate estimates of standing biomass from area-based surveys, showing marked variation in harvestable biomass of different corals. Further, limitations surrounding the development of *in situ* monitoring and biomass modelling capacity are discussed, highlighting important knowledge gaps, especially for species that occur in inter-reefal habitats.

Given limited data on stock status and trends, management assessments of Australia's coral harvest fisheries rely mostly on catch data combined with qualitative assessments of ecological riskOne strategy to transition towards a more defensible, quantitative, and ecologically relevant stock assessment framework relevant to Australian coral harvest fisheries is to develop a taxa specific biomass-based population model. However, the methodology, in addition to the practicality of attempting to establish a biomass-based model over an area the size of the 344,400 km^2 Great Barrier Reef Marine Park (which represents the bounds of the QCF) has not previously been examined in a commercial ornamental aquarium coral harvest context. To develop a biomass-based population model relevant to fishery operations, extensive biological and ecological data would be required to inform key modelling parameters. While such information does not currently exist for many LPS taxa, substantial data on growth, recruitment, and drivers of stochastic change already exists for *Acropora* spp. Therefore, using a combination of data already presented in this work, coral cover data provided by the Australian Institute of Marine Science Long Term Monitoring Program, in addition to various demographic and biological parameters provided (or converted) from other works in literature; an initial numeric biomass population model was developed for *Acropora* corals. This process, while not delivering a fit-for-purpose stock assessment model, established data inputs and management actions that are required for the development of more rigorous population models to inform fisheries stock assessments. To understand the influence of geographic scale, the same methodology was carried out across two defined geographic extents relevant to the QCF, (namely permitted zones) of the entire Great Barrier Reef Marine Park, as well as a defined region adjacent to Cairns. Similarly, using estimated biomass change values from geographically relevant reefs, a stochastic disturbance element was added to the numeric model (utilising Bayesian regression posterior distribution) to simulate the extreme stochastic change operating on *Acropora* populations through the defined model management area. This model highlighted the importance of incorporating ecological data to characterise the influence of stochastic stock dynamics for a genus such as *Acropora* that is strongly influenced by stochastic disturbance events.

Results of the research presented herein, show that there is a demonstrable need for the development of ecologically relevant, and most importantly, quantitative management capacity with Australia's commercial ornamental aquarium coral harvest fisheries. Historically, the large area of operation for Australia's major coral harvest fisheries and small scale and extreme selectivity of coral harvesting is presumed to minimise risks posed by sustained coral harvesting. However, escalating threats to reef ecosystems and documented declines in the health and abundance of hard corals necessitates improved understanding of the threat posed by coral harvesting. Failure to develop appropriate methods and undertake recurrent sampling to verify that commercial coral harvesting poses limited threat to targeted stocks and habitats will increasingly result in the restriction and exclusion of Australian harvested coral from major international markets. The work presented in this thesis contributes to improved understanding and management of Australian coral fisheries. For example, methods to assess coral biomass *in situ* for heavily targeted coral genera represents the necessary first step to relate harvest levels to stock size. While scientifically rigorous biomass-based population modelling is still out of reach, by developing an initial model, the potential data inputs required to generate an acceptable model have never been clearer. Ultimately, effective fisheries management and establishment of sustainable harvest limits will require considerable investment and ongoing research. However, Australia is uniquely placed to establish, and benefit from, demonstrated sustainability of commercial coral fisheries.

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Chapter 1

General Introduction

Tropical coral reef ecosystems, and extractive industries that rely on these systems, face unprecedented environmental and social pressure due to sustained and ongoing anthropogenic degradation (Lam et al., 2020; Pratchett et al., 2011). Coral reefs are highly diverse, abundant, and productive, providing many critical ecological goods and services (Hatcher, 1990; Moberg and Folke, 1999; Morais and Bellwood, 2020). Extractive industries, such as fisheries, provide food and livelihoods for many of the estimated 1.3 billion people inhabiting the tropics (Lam et al., 2020). However, these ecosystems are highly susceptible to the compounding influence of direct and indirect anthropogenic stressors, which may directly undermine fisheries productivity (McClanahan et al., 2014). Many anthropogenic activities directly impact coral reef environments, through sedimentation, eutrophication, pollution, and habitat degradation (Fabricius, 2005; Wear, 2016). Compounding this threat, the indirect influence of anthropogenically driven global climate change is likely to increase the frequency, intensity, and longevity of major disturbances, including tropical cyclones (Kang and Elsner, 2015; Sobel et al., 2014), and mass coral bleaching (Hughes et al., 2018b). Increasing environmental and habitat changes conflate the perennial and pervasive threat posed by fisheries exploitation (Pratchett et al., 2011; Rosenberg, 2003; Brander, 2009).

While large-scale and industrialised fishing represents a constant and continued threat to coral reef ecosystems (Jackson et al., 2001; Sadovy, 2005), there is also increasing evidence that small-scale, and often highly selective, fisheries may exert considerable pressure on target species and environments (Bruckner and Borneman, 2006; Knittweis and Wolff, 2010; Tissot and Hallacher, 2003; McClanahan et al., 2009). Widespread degradation of marine ecosystems is also increasing the imperative for environmental conservation (Worm et al., 2009), the objectives of which, often directly conflict with fisheries management (Jennings et al., 2014). Critically, reducing (if not eliminating) exploitation of functionally important marine species may be fundamental for improving ecosystem resilience (Bellwood et al., 2012; Bozec et al., 2016), enhancing resistance and/ or recovery in the face of increasing fishery-independent threats.

Ornamental aquarium fisheries extract a wide range of coral reef taxa, including reef fishes, corals and other invertebrates, for display in private and public aquaria (Wabnitz et al., 2003; Dee et al., 2014). The scale and extent of coral reef aquarium fisheries poses a considerable threat to heavily exploited and potentially vulnerable species (Pratchett, 2021a; Dee et al., 2014; Wood et al., 2012; Watson et al., 2023). Estimates of the number of corals entering the trade describe accelerating, though undulating, demand. Wabnitz et al. (2003) reports that around 934,400 pieces were harvested in 1997, rising to 1,142,200 pieces in 1999, before dropping to 942,661 pieces in 2001. Monticini (2010) states that 12 million pieces of coral are traded, composed of an estimated 140 species, though does not provide a time period. Tissot et al. (2010) states that 1.5 million live stony coral were removed (presumably) between 2004-2006. Rhyne et al. (2012) report that from 1990 to 2010, live coral pieces increased from an annual value of just under 200,000 to around 450,000 pieces. Using retailer surveys from 3 major coral importing nations, Watson et al. (2023) estimated that the annual total number of traded marine aquarium organisms equated to 55.4 million, without accounting for additional supply chain losses pre-sale. Some estimates suggest that marine aquarium fisheries are relatively high annual global value, being valued as high as US$ 2.15 billion (Watson et al., 2023). Fisheries supplying this market are typically highly selective hand-collection based operations, wherein particularly colourful or otherwise phenotypically notable specimens (often termed ‘rare’ in the trade) are in high demand and attract premium prices (Dee et al., 2014; Rhyne et al., 2014). Unlike fisheries targeting finfish or mobile invertebrates, coral harvest fisheries that supply the international ornamental aquarium coral trade have the potential to directly impact reef structure and function through direct removal of habitat-forming corals (Bruckner and Borneman, 2006; Knittweis and Wolff, 2010; Longenecker et al., 2015).

The majority of corals entering the international ornamental aquarium trade have been and continue to be harvested from the wild (Dee et al., 2014; Rhyne et al., 2012; Wood et al., 2012), although the wildharvest vs aquaculture contribution differs greatly by exporter and at a global level a growing proportion of aquarium corals are now originating from aquaculture operations (Morcom et al., 2018; CITES Wildlife TradeView, 2024). Some major exporters such as Indonesia have moved to the exclusive export of aquacultured corals only, while countries such as Australia remain dominated by wild harvested corals (source data available CITES Wildlife TradeView, 2024). Even within Australia, aquaculture is becoming more prominent, whether it be the more accesible fragmentation (i.e., asexual) based approach (applicaable only to some taxa, especially branching), or the development of commercially viable sexual reproduction based approaches (e.g., consider the award of) following the successful captive spawning of coral in recent years (e.g., Craggs et al., 2017, 2020; Puntin et al., 2023). There are also challenges associated with distinguishing corals that have been harvested from the wild versus those that are captive bred or otherwise produced through aquaculture or mariculture, and this has been exploited to misrepresent the sustainability of sustained coral exports from some countries (Wood et al., 2012; Chalias, 2021). Captive breeding has been undertaken for some aquarium corals, though the scale of production needs to be significantly increased to meet global demands (Craggs et al., 2017, 2020; Puntin et al., 2023). There will also be continued requirement to collect ‘brood stock’ from the wild. Asexual propagation (e.g., consider Sustainable Reefs, 2023) may therefore, be a more viable strategy in the short term to reduce reliance on wild stocks, but this strategy is still dependent on wild harvest to generate a variety and turnover of new products, particularly as the driving force for desirability in the hobby are new, unusual, and rare phenotypes. It is likely, therefore, that wild harvest will continue to be important to supply the extensive and growing marine aquarium trade (Dee et al., 2014; Rhyne et al., 2012; Wood et al., 2012).

1.1. International aquarium trade

Up until 2010, the major suppliers of marine aquarium corals were Indonesia and Fiji with the majority of corals destined for the USA and Europe (Rhyne et al., 2012; Wood et al., 2012). Contemporary trends in the nature and extent of the international marine aquarium trade are difficult to assess, largely due to deficiencies and extensive time lags in international trade reports as well as restricted accessibility to catch records (Pratchett, 2021a; Hughes et al., 2023; Tlusty et al., 2024). However, there have been significant recent events that are likely to have greatly affected patterns of world trade. Most critically, both Fiji and Indonesia imposed temporary bans on wild coral harvested in late 2017 (Adams, 2018; Chalias, 2019) due to socio-environmental concerns, such as deficiencies in local fisheries management and enforcement (Bruckner and Borneman, 2006; Knittweis and Wolff, 2010; Sosnowski et al., 2020). More recently (in 2021), the United Kingdom and then Europe banned the import of marine aquarium corals from Australia (European Commission, 2023) following noticeable increases in the level of coral exports despite extensive bleaching and widely reported declines in the health and condition of Australia’s coral reefs (e.g., Pratchett 2021a).

International trade in all hard (order Scleractinia) corals is regulated by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). CITES is a longstanding multilateral treaty with 184 signatory countries, which broadly aims to regulate international trade in order to prevent over-exploitation of endangered species (Ginsberg, 2002). International trade and corresponding harvest of these corals is, therefore, subject to national assessments and declarations that wild harvest fisheries do not undermine the population viability of target species, nor jeopardise the structure and function of relevant habitats and ecosystems (Wood et al., 2012). These conditions are often inferred to mean that approved fisheries for CITES species must be sustainable, though there is rarely the data or information needed to rigorously assess sustainability of coral harvesting (Harriott, 2003; Dee et al., 2014).

Aquarium fisheries have long been the subject of controversy and environmental concern, related not only to localised depletion of heavily harvested species (Tissot and Hallacher, 2003; Vagelli, 2008), but also detrimental fishing practices (e.g., cyanide-fishing, Barber and Pratt, 1998), and the introduction of non-native and potentially invasive organisms (e.g., Wiedenmann et al., 2001; Semmens et al., 2004). There is evidence that even low levels of harvesting can impact the abundance, population structure and viability of harvested coral species (Ross, 1984; Knittweis and Wolff, 2010), which is of particular concern given the ecological importance of corals and the increasing pressures facing coral reef ecosystems (Bruckner, 2000).

1.2. Australian coral fisheries

Australia has been and continues to be a major contributor to the international trade in aquarium corals, though its position relative to other major exporting countries is likely to fluctuate greatly due to global market influences (Wood et al., 2012, Rhyne et al., 2012, 2014). Following the sustained export cessation for Indonesia (and the temporary ban on export in Fiji), Australia became one of the most significant contributors to global trade simply by process of elimination. This is evidenced by the considerable jump in the number of wild *Acropora* coral pieces exported by Australia (according to importer reports) in the year prior to the ban (80,779) compared to the years immediately following the ban (144,016 and 149,740 for 2018 and 2019 respectively; CITES Wildlife TradeView, 2024).

Most coral exported from Australia is harvested from within the Great Barrier Reef Marine Park and World Heritage Area by the Queensland Coral Fishery (QCF). Up until the 1980s commercial coral harvesting on the GBR was focussed on fast growing branching corals (Acroporidae and Pocilloporidae) that were sold mainly as curios, or souvenirs (Harriott, 2001). In the late 1980s, however, there was a rapid shift towards harvesting of live corals for the marine aquarium industry (Harriott, 2001), which led to marked shift in the types of corals harvested.The Total Allowable Commercial Catch (TACC) for the QCF (200 tonnes) is split between "specialty corals" (60 tonnes) and "other corals" (140 tonnes), whereby the latter includes Acroporidae and Pocilloporidae corals, which were originally harvested as curios (McCormack, 2005).

Most corals harvested exclusively for the marine aquarium industry ("specialty corals") have large, fleshy polyps (e.g., Lobophyllidae), which are considered to be relatively slow-growing and therefore, potentially vulnerable to over-harvesting (Harriott, 2003). Many of these species are also collected from inter-reefal habitats (Pratchett et al., 2020a, 2020b). While Acroporidae corals now dominate both aquarium and curio harvests for the QCF , the retention of the original quota split (30:70), has been justified on the basis that Acroporidae and Pocilloporidae are fast growing and should therefore, be treated separately from "specialty corals" (e.g., Atkinson et al., 2008). This effectively allows for up to 200,000 kg of live coral (and up to 140,000 kg of Acroporidae) to be harvested each year by the QCF.

Historically, the QCF has been presumed to pose limited threat to wild coral stocks mainly because harvest levels and limits were considered trivial compared to the amount of coral that exists across the broad expanse of the GBR (Oliver, 1985; Harriott, 2001; Atkinson et al., 2008). However, there is increasing evidence that coral harvesting is disproportionately focussed on very specific and sometimes rare species (Militz et al., 2018), with concentrated fishing in limited areas where these corals occur (Jones, 2011; Pratchett,2021a). Similarly, when quotas are set at the genus level, there is heightened potential for overexploitation of selected rare species under legal established quota (Rhyne et al., 2012). Sustained harvesting of corals is also potentially compounded by increasing incidence of major disturbances, and corresponding declines in the size and abundance of corals on the GBR (Hughes et al., 2018a; Dietzel et al., 2020; Pratchett, 2021). Despite espousing the QCF, Harriott (2001) stated that significant and ongoing research was needed to establish status and trends in population size and viability, and thereby justify continued exploitation of heavily harvested and potentially vulnerable coral species.

Coral harvest fisheries may represent a familiar scenario to managers and scientist working in data-depauperate and/or invertebrate fisheries (Anderson et al., 2011; Chrysafi and Kuparinen, 2016; Fröcklin et al., 2014). Management of Australia’s coral harvest fisheries is heavily reliant on catch data, combined with qualitative assessments of ecological risk (e.g., State of Queensland, 2021). There are certainly existing qualitative and semi-quantitative methodologies that have been (e.g., ERA; Pratchett, 2021a) or could be (e.g., productivity-susceptibility analysis, applied to other ornamental fisheries; Baillargeon et al., 2020) applied to data-poor operations such as coral harvest fisheries. However, when considering the seemingly heightening quantitative data expectations needed to provide evidence for non-detrimental harvest to the satisfaction of the global community (European Commission, 2023), wild-caught based coral harvest fisheries are likely to require the development of quantitative management strategies in the near future.

Under present management strategies for the QCF, corals ascribed to different risk categories (or tiers) are subject to differential constraints that are based on relative recent (2016-2017 through to 2018-2019) reference harvest levels (State of Queensland, 2021). However, decision rules proposed in the QCF 2021-2026 Harvest Strategy (State of Queensland, 2021) were not consistently applied across corals in different risk categories, and also allowed for further increases in harvest levels of many species for which there have been largely unconstrained and unregulated increases up until 2018-2019 (Pratchett, 2021a). Much more restrictive harvest limits were therefore, imposed by the Australia's CITES Scientific Authority as part of the necessary endorsement for coral exports. Meanwhile, significant recent increases in coral exports (and corresponding harvest levels) by the QCF (and Australian coral fisheries, more generally) also attracted the attention of the European Scientific Review Group (SRG) on CITES, which subsequently imposed bans on the import of several coral species (*Duncanopsammia axifuga*, *Catalaphyllia jardini*, *Fimbriaphyllia ancora*, *Fimbriaphyllia paraancora*, and *Euphyllia glabrescens*) from Australia (Gay, 2022). Similarly, the Joint Nature Conservation Committee (JNCC) in the United Kingdom raised concerns about the sustainability of escalating exports of Queensland corals, which led to a temporary ban on the import of all corals harvested by the QCF (Gay, 2023).

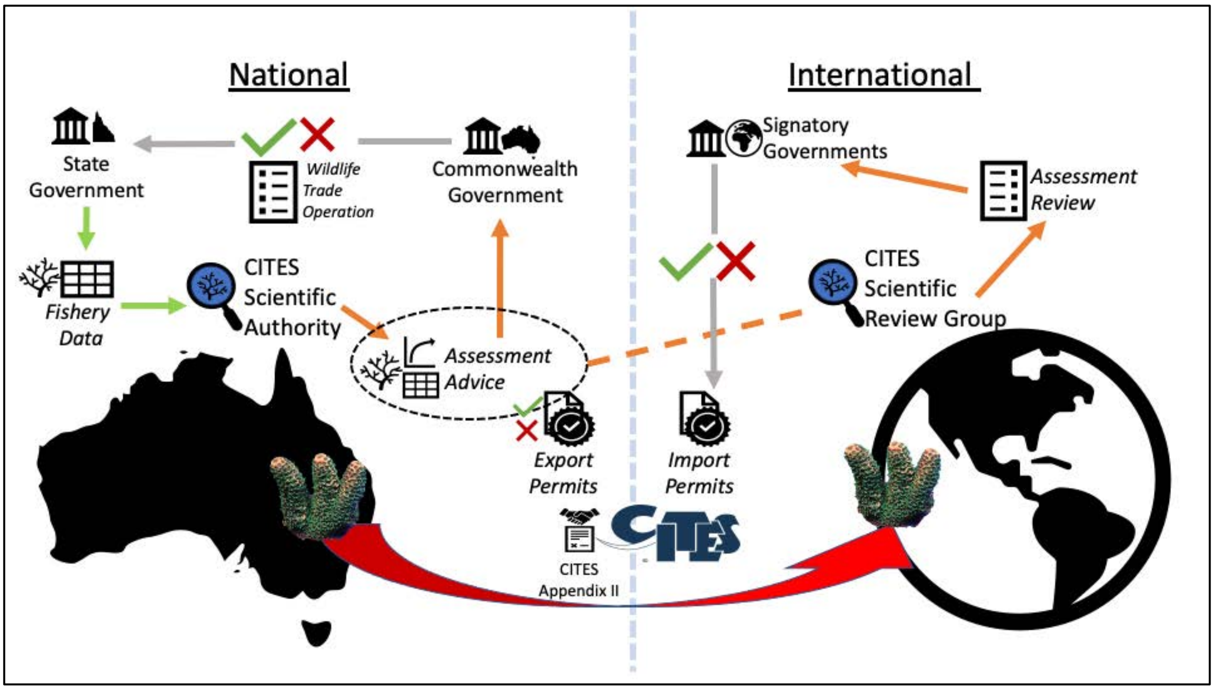

Figure 1.1. Diagram depicting the complex export permit approval and revision process in Australia. Export of corals ultimately depends on the periodic re-approval of coral harvest fisheries for each state as a Wildlife Trade Operation, as issued by the Australian Federal Commonwealth Government. The CITES Scientific Authority will organise the independent scientific assessment of the sustainability of trade. State governments are responsible for collecting the key fishery data required to provide evidence for approval. Scleractinian corals are Appendix II listed organisms, meaning that evidence must be provided to assure that the trade of targeted species does not impact their survival in the wild (i.e., an NDF or Non-Detriment Finding). Being Appendix II listed organisms, scleractinian coral collected in Australia can only legally be exported with the issuance of a CITES export permit, and imported with a CITES import permit. Given the CITES process, it is also possible for the Scientific Review groups of other participating nations to recommend targeted bans on the import of corals in instances where concerns arise as to sustainability or supporting evidence of NDF.

Even though overall harvest levels of hard corals for the QCF are still well within legislated and sustained catch limits (e.g., 60,000 kg for specialty coral species), there is a recognised need to constrain increasing harvest levels of heavily harvested coral species (e.g., DEEDI, 2009; DAF, 2021). However, sustained and significant increases in harvest levels alongside increasing fishery-independent threats (Hughes et al. 2018b) and widely reported declines in coral cover across the GBR (Mellin et al., 2019) has brought about significant pressure to review and reform management of Australian coral fisheries.

1.3. Differential vulnerability to harvesting

Corals vary greatly in their vulnerability to fisheries exploitation, as well as increasing fishery-independent threats (Pratchett et al. 2020a, 2020b). This differential vulnerability to harvest is owed to marked variation in their physiology and demography. Such differences are exemplified by contrasts between small polyp (SPS) versus large polyp (LPS) species of hard corals (Bruckner, 2003). SPS corals (e.g., Acroporidae) are characterised by fast growing, mostly branching coral species (Bruckner, 2003). These corals are considered to be very resilient to harvesting, owing to their high abundance and exceptional capacity for recovery (Harriott, 2001; Bruckner, 2003); SPS corals are disproportionately affected by most major disturbances, including climate-induced mass bleaching (Hughes et al., 2018b), cyclones (Madin, 2005; Madin and Connolly, 2006), and population outbreaks of crown-of-thorns starfish (Johansson et al., 2016), yet their persistence and predominance in most shallow reef habitats is testament to their exceptional capacity for growth and replenishment (Pratchett et al., 2020c).

One of the foremost constraints to effective management of SPS corals are challenges associated with their taxonomy and nomenclature (Ball et al., 2021; Bridge et al., 2023; Cowman et al., 2020). Notably, *Acropora* is the most diverse genera of Scleractinia, accounting for >30% of all reef-building corals (Madin et al., 2016). However, the difficulties in distinguishing different species are apparent based on current taxonomic revisions (Bridge et al. 2023), and reflected in allowances for export levels of many different SPS corals (including *Acropora*, *Montipora*, *Pocillopora*, *Stylophora*, and *Porites*) to be reported at the level of genus (CITES, 2016). It is nonetheless, important to understand the species-specific catch composition within these genera, especially given species-specific differences in their distribution, abundance and vulnerability to harvesting (e.g., Wabnitz et al., 2003). Given the extraordinary diversity in the size and growth of SPS corals, there are also challenges in effectively quantifying population status and trends (e.g., Bruckner and Borneman, 2006; Knittweis and Wolff, 2010). Critically, SPS corals may form very large colonies, which are often fragmented during harvesting and disturbances (Rhyne et al., 2012; Roth et al., 2013). This capacity for asexual reproduction is such that coral fishers may collect whole colonies of coral (which are often fragmented post-cleaning back at the facility), or may fragment *in situ* individual pieces. This has a number of important ecological and harvest related implications. Firstly, it could be assumed that the collection of fragments may be ecologically favourable, as this may allow retention of greater biomass compared to the removal of entire colonies. Secondly, the collection and sale of both colonies and fragments (which can have vastly different weights and dimensions) requires that coral be considered in terms of both numbers (as recorded on CITES export and import permits), as well was weight (as harvest quota is recorded). These two metrics must be used to elucidate trends fully, as an increase in numbers may not indicate an increase in weight (and vice versa).

LPS corals (e.g., *Euphyllia* spp.) were the traditional mainstay of marine aquarium fisheries and continue to represent a significant component of the global trade in aquarium corals (Rhyne et al., 2012). The historical focus on LPS corals is attributable to their comparative amenability to harvest and transport (though colloquially, even this is said to be highly variable and species and/or harvest location dependent), and relative ease of care in aquaria (Rhyne et al., 2012). Like SPS taxa, aquaculture and mariculture operations continue to grow as a source of LPS corals; although as some taxa are reported to be untenable for asexual fragmentation, these taxa are likely to remain and/or grow in terms of wild harvest contribution to trade at least for the near future (Rhyne et al., 2014). However, it is generally assumed that LPS corals are slow growing and long-lived, and are therefore, much more vulnerability to sustained and significant harvest pressure (Bruckner, 2003; Harriot, 2003). Accordingly, Bruckner and Borneman (2006) estimated that sustained harvesting of LPS corals in Indonesia had reduced the standing stock by 96%. Knittweis and Wolff (2010), meanwhile, demonstrated significant effect of harvesting on the abundance and size structure of the solitary coral *Heliofungia actiniformis*. Demonstrating significant effects of harvesting on LPS corals may however be much easier than for SPS corals, as many are monocentric (occur as single polyps) and tended to be harvested in their entirety (rather than fragmented). Population status and trends may therefore, be readily inferred from patterns of abundance (Knittweis and Wolff, 2010) or occurrence (Bruckner and Borneman , 2006). Information on the size or weight of corals (both in terms of what is harvested and what is available) is nonetheless important for establishing fisheries impacts and improving management, especially given weight-based quota limits for Australian coral fisheries (e.g., McCormack, 2005).

1.4. Thesis objectives and structure

The purpose of the research presented in this thesis is to redress (at least in part) critical information and data gaps that undermine and constrain contemporary management of Australian coral fisheries, with a major focus on the QCF. The thesis comprises four separate research chapters, each with distinct objectives:

Chapter 2 examines catch data from QCF across 14 successive annual fishing seasons from 2006-2007 up until 2019-2020. The primary goal of this research was to quantify the overall level and annual rate of increase in reported harvest levels for two SPS (*Acropora* spp., *Montipora* spp.) and nine LPS taxa (*Acanthophyllia deshayesiana*, *Catalaphyllia jardinei*, *Cycloseris cyclolites*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Fimbriaphyllia ancora*, *Homophyllia* cf. *australis*, *Micromussa* cf. *lordhowensis*, *Trachyphyllia geoffroyi*). These data are important for assessing the appropriateness of historical management arrangements (e.g., McCormack, 2005) as well as recent fisheries reforms (DAF, 2021), intended to prevent unconstrained and unregulated growth in harvest levels of individual species or genera. While the QCF has overarching quota limits for specialty corals (60 tonnes) and other corals (140 tonnes), there are no legislated limits on the harvest of individual coral species or genera. It is expected that historical trends regarding the growing proportion of SPS coral taxa in global trade (Rhyne et al., 2012; Wood et al., 2012) will be reflected in Queensland Coral Harvest over the referenced period.

Chapter 3 seeks to establish size-weight relationships for relevant growth forms of *Acropora* spp. While there is extensive data on the size and abundance of *Acropora* corals on Australia's Great Barrier Reef (e.g., De'ath et al., 2012; Dietzel et al., 2020), few studies record weight or biomass of these corals; the only relevant data on size-weight relationships of *Acropora* spp. comes from limited sampling of just one growth form in Papua New Guinea (Longnecker et al., 2015). In this study, size-weight relationships were based on data recorded for a total of 3,972 corals across six different growth forms (bottlebrush, caespitose, corymbose, digitate, staghorn and table). These data were then combined with video transects (n=41) to calculate harvestable biomass of *Acropora* spp. at a range of sites across the GBR. The coloniality of *Acropora* corals poses a management challenge from a quantification perspective, the capacity for fragmentation *in situ* during collection may have ecological benefits. It is hoped that the generation of growth form specific size-weight relationships will shed light on potential best harvest practices for the harvest of *Acropora* corals in the Queensland Coral Fishery.

Chapter 4 modelled size-weight relationships for six different LPS coral species: *Catalaphyllia jardinei*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Homophyllia* cf. *australis*, *Micromussa lordhowensis*, and *Trachyphyllia geoffroyi*. Aside from facilitating estimates of standing biomass for these heavily harvested coral species, establishing size-weight relationships provides an opportunity to test whether these corals exhibit isometric or allometric growth. LPS taxa, particularly monostomatous inter-reefal taxa have typically been regarded as those most vulnerable to direct fisheries exploitation. It is expected that the description of size-weight relationships for these taxa may enable consideration of the value of potential minimum and maximum size limits.

Chapter 5 presents an initial biomass population model for *Acropora* corals, which is used to simulate stochasticity and explore the probability of population collapse across a broad range of different levels of fisheries exploitation, and helps to establish data inputs and management actions that are required for the development of more rigorous population models to inform fisheries stock assessments. Given the ecological importance of *Acropora* spp., increasing harvest levels and the mounting fishery-independent pressures, quantifying the impact of fishing within an ecological (ideally stochastic) and probabilistic context is a major priority. This is doubly true when expectations regarding the level of quantification required for non-detriment findings appear to be increasingly enforced through import bans internationally (European Commission, 2023). It is expected that, if a initial biomass-based population model for *Acropora* at the current spatial resolution of management can be constructed, it is likely to result in considerable biomass estimates across the entire harvestable Great Barrier Reef Marine Park Area. How realistic this representation is in reality, considering factors such as spatial concentration of harvest, as well as taxonomic resolution of targeted *Acropora* taxa, is unclear.

Wide-spread and accelerating degradation of coral reef systems is increasing environmental and political pressure on the QCF and other coral fisheries, both nationally and internationally. Due to the consequences of inaction (i.e., restriction of market access) environmental and economic needs are in this case aligned with the development of a quantitative harvest strategy. Failure to ensure the sustainability of these industries threatens ecological impacts such as alteration of population demography, loss of abundance, and loss of genetic diversity over unknown (though most likely local) scales. Similarly, due to the status of Scleractinia as Appendix II listed organisms, failure to monitor and manage any ecological consequences of harvest will result in restriction of market access and eventual economic strangulation of Australian coral harvest fisheries.

Chapter 2

Harvest trends in the Queensland Coral fishery from 2006-07 to 2019-20

2.1. Introduction

Effective fisheries management is a key strategy necessary to promote the resilience of threatened marine environs, such as coral reefs, through recognition of the functional importance of target taxa (Bellwood et al., 2012; Bozec et al., 2016). Unmitigated acceleration and expansion of fisheries exploitation is among the most prominent anthropogenic threats to coastal ecosystems (Bellwood et al., 2019; Fenner, 2012; Hawkins and Roberts, 2004; Pandolfi et al., 2003; Roberts, 1995). The depletion of heavily harvested species, as well as environmental degradation, resulting from destructive large-scale industrialised fishing has long been a major concern for marine ecosystems (Berkes et al., 2006; McManus et al., 1997; Pauly et al., 2005). However, the impact of sustained small-scale fisheries (e.g., subsistence and artisanal fisheries) may also pose a significant threat to targeted species and habitats (Alfaro-Shigueto et al., 2010; Russ, 2002; Wells, 1997). The fisheries operating in tropical coral reef environs tend to focus on the harvest of a diverse array of multi-species finfish or mobile marine invertebrates food stocks (Melnychuk et al., 2017; Robinson et al., 2020). Occurring in conjunction with wide-scale degradation of coral reef ecosystems (Hughes et al., 2017a), the exploitation of these animal populations is expected to compound upon the co-occurring impacts of habitat loss (Lam et al., 2020; Wilson et al., 2008). In contrast, coral harvest fisheries for the ornamental marine aquarium industry have the potential to directly affect the function and physical structure of coral reefs and adjacent inter-reefal habitats through the direct removal of habitat-forming corals (Bruckner, 2000; Daley and Griggs, 2008).

The ornamental marine aquarium industry is a multi-billion dollar industry that facilitates the international trade of millions of live animals (i.e., live fish, corals, and other invertebrates) each year (Linardich et al., 2024; Rhyne et al., 2017, 2012; Wood et al., 2012). The environmental reputation of the fisheries supplying the international aquarium trade has continued to raise concerns globally, following the use of highly destructive fishing practices (e.g., cyanide fishing; Barber and Pratt, 1998) in developing nations, as well as enabling the introduction of exotic and potentially invasive marine organisms (Semmens et al., 2004; Wiedenmann et al., 2001). The potential profitability of small-scale fisheries supplying the international aquarium industry is such that there have historically been severe reductions of species abundance in marine systems for targeted fish species (e.g., *Zebrasoma flavescens* and *Pterapogon kauderni*; Tissot and Hallacher, 2003; Vagelli, 2008), although with the implementation of strategic management, these fisheries can recover and populations can be sustainably harvested (as occurred for previously exampled fish species; see Rhyne et al., 2017). Wild caught coral harvest fisheries supplying this market are no exception in terms of potential overexploitation in the absence of management (or incorrect or inefficient management), and establishing and maintaining sustainable harvest practices has been a consistent struggle internationally (Bruckner and Borneman, 2006; Harriott, 2001; Longenecker et al., 2015). This can be exemplified by a history of international import and export bans applied to and/or by major exporting nations (at least in part) as a response to environmental concerns, whether scientifically or politically motivated (e.g., Chalias, 2019).

All hard corals (order Scleractinia) are listed in Appendix II of the Convention on the International Trade of Endangered Species of Flora and Fauna (CITES), meaning that for export approval, evidence that harvest does not detrimentally impact target populations must be supplied (i.e., non-detriment finding; NDF). In the absence of appropriate evidence, export can be suspended to all 184 signatory nations, and otherwise considered illegal trade (Petrossian et al., 2020; Sosnowski et al., 2020). Similarly, nations may also suspend import from particular countries if their residing scientific review group does not find the supplied evidence to support NDF sufficient proof of non-detriment; as was recently the case in the UK, which actioned a ban against all corals originating from Queensland, Australia (European Commission, 2023; Gay, 2022). Given the domestic market is typically very small in harvesting nations such as Australia, restriction of access to the international markets threatens the economic sustainability of coral harvest fisheries. Therefore, maintaining the global reputation of sustainable harvest by updating management systems to reflect threat to target species, broader environmental impact, and scale of harvest is particularly key for coral harvest fisheries.

Australia's practices for managing coral harvest fisheries have historically been considered as a leading example of sustainable management of wild coral harvest internationally (Harriott, 2003; Wood et al., 2012). Australia was estimated to contribute only ~4.5% of corals in global trade between the period of 2000 to 2010, with the primary contributors being Indonesia (70%) and Fiji (10.3%); while major importing nations were the Unites States of America (US) importing 61% of global trade, and European countries accounting for 31% (Wood et al., 2012). Rhyne et al. (2012) stated that despite Australia's relatively small contribution to global trade, imports into the US had been increasing over the same period (2000-2010), noting that Australia had gained a reputation for more expensive, higher quality coral exports. Australia has three major state-run coral harvest fisheries, in Queensland (Queensland Coral Fishery; QCF), the Northern Territory (Northern Territory Aquarium Fishery), and Western Australia (Western Australian Marine Aquarium Fish Managed

Fishery). The QCF is the largest of Australia's coral harvest fisheries with a total contemporary harvest quota of 200 t, and a spatial extent spanning harvest permitted zones of the 344,400 km^2 Great Barrier Reef Marine Park (Pratchett, 2021a). It is also worth noting that harvest quota at the state level is managed in terms of total weight (t), while CITES records live coral pieces for international ornamental aquarium coral import and export by the number of coral pieces. Considered generally (i.e., across taxa), coral pieces are highly irregular in terms of both size, shape, and weight; and can represent whole coral colonies, or colony fragments. For this reason, weight and/or number of pieces alone tells only part of the story as each measure is required to understand otherwise masked trends within the other. For example, number of pieces is required to reveal if there has been a change in the size of coral pieces harvested over time. Understanding these trends is important, as a higher number of smaller coral pieces (that is of equivalent weight) could have implications for various fisheries metrics (e.g., CPUE) as well as ecological impact of harvest. The QCF has always used input (limited licenses) and output (annual harvest) controls (Harriott, 2001). Historically, the potential harvest quota of the QCF has been considered to be much less than 1% of total available coral biomass, and thus a sustainably managed fishery (Harriott, 2001). However, this estimation was made when the fishery operated within 50 discreet zones, each tied to an individual harvesting license with a limit of 1 t per zone (50 t quota total). The average size of these harvest zones was 25,000 m^2, and the total area of the park open to fishing was less than 1% (Harriott, 2001). It is worth noting that even in this historic assessment work, uncertainty about appropriate harvest levels for particular species was identified, along with a need for high taxonomic resolution in recording data (Harriott, 2001). In comparison to historic zone-based management, contemporary management strategies came into effect with the establishment of the 'Great Barrier Reef Marine Park Zoning Plan 2006', and implemented a roving license approach, with fishing open in permitted zones to commercial license and quota holders (Pratchett, 2021a). Total allowable commercial catch was increased to 200 t and divided into a 60 t live ‘specialty coral’ category for ‘large polyp stony’ (LPS) corals, and a 140 t ‘other coral’ category for ‘small polyp stony’ (SPS) species.

The inequitable 30/70% division between LPS and SPS taxa in the QCF reflects the historic focus on the harvest of branching SPS taxa such as Acroporidae and Pocilloporidae as curios and live rock (i.e., not as living specimens), as well as the presumed minimal harvest risk to these taxa (Pratchett, 2021a; Harriott, 2001) in light of high abundance and wide (though patchy) distribution. As aquarium technology has continued to improve, keeping of reef-building SPS genera such as *Acropora* spp. and *Montipora* spp. has become more accessible, and so ‘mini-reef’ tanks featuring SPS taxa have become increasingly desirable among marine aquarium hobbyists (Dee et al., 2014; Rhyne et al., 2012). This shift is part of the historic progression from the keeping of only marine aquarium fish, to combinations of fish and easy to keep invertebrates (including cnidaria such as soft corals), towards the keeping of LPS and finally SPS taxa (Murray et al., 2012; Rhyne et al., 2009). In the period from 2000 to 2010, SPS Acroporidae corals formed a major and growing proportion of international live coral harvest at an estimated 363,121 pieces according to exporter reports (Wood et al., 2012). Being major reef-building taxa, concerns for these species have often been overshadowed by the potential for over-exploitation of the comparatively less abundant and slower-growing LPS taxa (Bruckner and Borneman, 2006; Rhyne et al., 2012). *Acropora* spp. and similar SPS taxa are typically fast-growing, and being both colonial and branching, are potentially resilient to harvest depending on prevailing practices, as fragmentation can be conducted *in situ* leaving part of the colony intact and reducing the overall environmental impact (Chapter 3; Longenecker et al., 2015; Pacey et al., 2022). However, recent work on coral harvest in the QCF has highlighted the potentially cumulative and exacerbating nature of harvest in light of vulnerability to large-scale disturbances (Pratchett, 2021a; Pratchett et al., 2020a). Major SPS taxa such as *Acropora* spp., are often highly constrained by major disturbances such as cyclones and tropical storms, coralivorous predator irruptions (e.g., crown-of-thorns starfish), as well as mass coral bleaching events (Pratchett et al., 2013; Wakeford et al., 2008). Additionally, harvest is highly selective, targeting only specific phenotypes (which may or may not represent species), and spatially constrained (at least relative to the Great Barrier Reef Marine Park), although spatial data from the fishery is difficult to access. Given the likelihood of increased occurrence and intensity of at least some of these events (Cheal et al., 2017; Hughes et al., 2017b; Kang and Elsner, 2015; Sobel et al., 2014) as a result of global climate change, it is unclear whether contemporary management has the capacity to deal with the growing popularity of these taxa, or recurrent environmental threats external to fishery pressure.

LPS have typically been set apart from SPS taxa by presumed direct harvest risk, as exemplified by contemporary QCF harvest limits (i.e., 60 t; 30/70% split of 200 t). Historic concern for overexploitation of LPS species is well founded (Harriott, 2003; Rhyne et al., 2014), and has been a recurrent issue in major coral harvesting nations, such as Indonesia (Bruckner and Borneman, 2006; Knittweis and Wolff, 2010). As the name implies, LPS species tend to have large, fleshy polyps. This group (i.e., LPS corals) is typically perceived by hobbyists as easier to maintain in aquaria compared to many SPS taxa, and LPS coral taxa were the most accessible coral type prior to the improved accessibility and precision of aquarium hobby technology (Rhyne et al., 2012; Wood et al., 2012). Though, it is important to distinguish that LPS corals have not been supplanted by SPS taxa, and many high and low value taxa remain highly desirable in their own right (e.g., Helgason, 2020). These species are associated with slow growth, at least relative to branching and reef-building taxa (Pratchett et al., 2020a). Popular species often have extended daytime tentacles (Pratchett et al., 2020a, 2020b) with a wide range of phenotypic forms, including tentacled flabello-meandroid species such as *Euphyllia glabrescens*, *Catalaphyllia jardinei*, and *Fimbriaphyllia ancora*; sub-massives such as *Micromussa lordhowensis*; and monostomatous or 'solitary' single polyp species such as *Cycloseris cyclolites*, *Acanthophyllia deshayesiana*, *Trachyphyllia geoffroyi* and *Homophyllia australis*. Solitary species are believed to have particularly slow growth, which the limited available research suggests may be highly stochastic and difficult to measure *in situ* (Pratchett et al., 2020a).

The aforementioned LPS species, as well as many other target LPS species are inter-reefal, and are most commonly collected from aggregations in soft-sediment habitats (Bruckner and Borneman, 2006; Pratchett et al., 2020a; Rhyne et al., 2012, 2014). Compared to major reef-building taxa such as *Acropora* (Ball et al., 2021), these species appear less commonly in scientific studies and monitoring surveys, which are more typically conducted on contiguous coral reefs. Due to a lack of prior research, key details pertaining to growth, recruitment, reproduction, abundance, and distribution are missing for many of these heavily targeted taxa (but see Pratchett et al., 2020a). The continued ambiguity surrounding key biological and ecological information greatly inhibits the capacity for management to determine the sustainability of the growth in reported coral harvest over the last 15 years highlighted during the most recent approved wildlife trade operation assessment (Commonwealth of Australia, 2021; Pratchett, 2021a).

Australia's reputation for well managed coral harvest fisheries appears to have diminished in more recent years. While something as intangible as 'global reputation' is difficult to measure, perceived reputation may be a valuable commodity on the world stage for delegates attempting to navigate the complex politics of multi-lateral agreements such as CITES. The UK has not been the only nation to implement a trade ban on Queensland wild caught corals, with the EU also implementing a trade ban on harvest fisheries (European Commission, 2023; Gay, 2022) citing insufficient supporting quantitative data in the most recent wildlife trade operation assessment (Commonwealth of Australia, 2021; Pratchett, 2021a). For the first time in the QCF's history, 11 coral taxa (2 genera, 9 species) received species specific harvest quotas in an attempt to limit considerable growth in coral harvest (Commonwealth of Australia, 2021; Pratchett, 2021a). Coral harvest in Australia has multiple levels of regulation, from state, to federal, to international regulation of trade via CITES (Pratchett et al., 2020a). While no exceedance of harvest quotas would be assumed to be indicative of sustainable harvest, in reality, sustainable harvest can still be threatened by incorrectly set quotas (such as ecologically uninformed quotas), by the acceleration of harvest even within prescribed quotas (especially where threshold catch trigger values have not been described), or by failure to effectively implement and/or follow prescribed management strategies (Beddington et al., 2007; Rhyne et al., 2009). This included two SPS taxa *Acropora* spp., *Montipora* spp., and nine LPS taxa *Acanthophyllia deshayesiana*, *Catalaphyllia jardinei*, *Cycloseris cyclolites*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Fimbriaphyllia ancora*, *Homophyllia* cf. *australis*, *Micromussa* cf. *lordhowensis*, *Trachyphyllia geoffroyi* (Commonwealth of Australia, 2021; Pratchett, 2021a). This study further examines harvest trends highlighted in the most recent WTO assessment (Pratchett, 2021a) temporally and by catch composition, for overall harvest, family, and selected species of interest. Further, contemporary trends, the implications of these trends in light of historic harvest, and management approach shifts post- wildlife-trade assessment will be explored within the context of key taxa identified during the latest WTO assessment (Commonwealth of Australia, 2021), while considering the requirements for future actions and the wider implications for management of coral harvest fisheries in Australia and globally. In the current global landscape, failure to provide quantitative evidence for non-detriment findings, as well as fluidity and self-critical assessment of employed management strategies is likely to result in a diminishing international opinion of Australian coral harvest fisheries and further market access restrictions; which poses a serious threat to the continuation of these fisheries.

2.2. Methods

All reported harvest level data was provided by the Queensland Government Department of Agriculture and Fisheries (DAF) on May 17th 2021, which included all available information for estimated and recorded weights. The dataset supplied by the Department of Agriculture and Fisheries consisted of a total of The data is collected from fishery logbooks by the Queensland Department of Agriculture and Fisheries, who determine reporting categories and appropriate levels of data aggregation. Analyses was conducted at both the family level and at the species level according to available genus or species level categories. A family level category was generated by assigning a family to all provided 'ReportName' categories, updated to reflect current taxonomy according to the online database World Register of Marine Species (WoRMS; Costello et al., 2013; Horton et al., 2017). The same updating procedure was then followed for the species level analysis (i.e., using WoRMS). All data processing and visualisation was conducted via the opensource software environment R version 4.2.2 (R. Core Team, 2022), with additional packages noted where applicable.

2.2.1. Catch composition

Catch composition (%) plots at family and species level were produced by summarising the total number of pieces for each group for the 2019-2020 financial year (i.e., the most recently assessable financial year) with full data available.

2.2.2. Select species retained pieces and weight temporal trends

Following the most recent assessment of the QCF, 11 taxa (two genera, 9 species) were placed under taxa specific harvest limits for the first time in order to address significant increases in harvest accompanied by failure to initiate management strategy responses to defined catch trigger thresholds during contemporary management (Commonwealth of Australia, 2021; Pratchett et al., 2020a). Temporal trends for these taxa will be further examined at the genus/species level. GAMs are commonly used to examine trends in data, and simple iterations of this methodology are particularly well suited to instances where these is no need to establish relationships between variables, and where patterns examined are likely to be non-linear (Wood, 2011). To represent catch trends in both retained number of pieces, as well as retained (and in the absence of species level retained weight) estimated weight for these species, General Additive Models (GAMs) were fit using the formula 'y ~ s(ReportYear, k = 3', with splines (k) via the package 'mgcv' (Wood, 2011) limited to 3 due to the maximum data points available for some groups, and a maximum of 14 points representing financial years between 2006-2007 and 2019-2020.GAMs were visualised using the 'geom_smooth' function of the ggplot2 package (Wickham, 2016).

When visualising retained pieces, additional grouping variables were created to identify instances of grouping category change (i.e., 'pre-split' and 'post-split') data for these, typically implemented in response to changes to species taxonomy and/or improvements in taxonomic recording resolution through time in retained pieces. These pre- and post-split groups were created based on initial plotting of 'Retained Number' values summarised to year for each unique taxa category using a loess smoother (Wickham, 2016) combined with comparison of current and historical taxonomy. As an illustrative example, a decline in the '*Euphyllia* spp.' ReportName category coincided with the appearance of *Euphyllia ancora* in 2016. This species has since been assigned to the genus *Fimbriaphyllia* (Luzon et al., 2017), suggesting that *F. anchora* was likely recorded under the *Euphyllia* spp. category prior to 2017-17 financial year.

To allow assessment of select genus/species level temporal trends for weight (kg), the unit in which total allowable commercial catch (TACC) is set, an average weight value (Appendix A, Fig A1) was used to estimate weight based on total retained pieces. To determine the individual weight of corals for the period between 2010-2011 and 2015-2016 financial years, the annual reported weight (in kg) was divided by the total number of retained pieces for each genus/species. For all corals harvested as specialty corals, the individual weight of corals was then multiplied by 0.75, based on the agreed allowance for offcuts (Pratchett and Messmer, 2017), thereby better reflecting the relevant weight of these corals that is reported against the specialty coral quota in 2016-2017 through 2019-2020 (DAF, 2016). GAMs were used to visualise the trend (increasing, decreasing, flat or unclear pattern) for each group in the weight per piece values for each year (between 2010-2011 to 2015-2016 financial year) following the method described above (see Appendix A, Fig A1). For select genus/species level plots pre-split data were used for this estimation where necessary and considered appropriate. In the absence of weight per-piece data, a value from a similar species was substituted to allow estimation (e.g., the average value for *Cycloseris cycloites* was used in estimation for *Acanthophyllia deshayesiana*). GAMs were fit to the retained weight (kg) and (where necessary) estimated weight values to indicate the total weight trend over time. These estimated weights were also compared to direct measurements of the average weight of live corals, recorded by Pratchett and Messmer (2017), based on weighing of 7,418 individual coral colonies or pieces collected by the QCF.

2.2.3. Total retained pieces and retained and estimated weight through time

The total number of retained pieces for each financial year was plotted through time, and then connected using a simple line graph (Wickham, 2016) to demonstrate trend from year to year. Estimated weight was calculated by multiplying the number of pieces by average weight per species or family to account for weight data from 2016-17 onwards. See Appendix A, Fig A1 for all utilised values. Linear models were fit to defined sections of the data (2006-07 to 2016-17; 2016-17 to 2019-20; 2017-18 to 2019-20) to asses the slope of each defined section, and in the case of retained and estimated weight, assess the similarity of coefficients following methodology outlined by Clogg et al. (1995).

2.3. Results

2.3.1. Total retained pieces and retained and estimated weight through time

Coral harvest has increased >750% from the 2006-07 to 2019-20 financial year for total retained hard coral (Scleractinia) pieces (Figure 2.1a), going from just over 56,000 pieces at the implementation of contemporary management arrangements in 2006-07, to over 490,000 retained pieces in 2019-20. The trend for retained pieces is reasonably steady from 2006-07 to 2017-18 increasing at a rate of 21,675 per year (Linear Regression; β estimate = 21674 ± 848 S.E.), with a sharp increase in slope from the financial years 2017-18 to 2019-20 (Linear Regression; β estimate = 94922 ± 23570 S.E.) increasing at a rate of 94,922 per year (Figure 2.1).

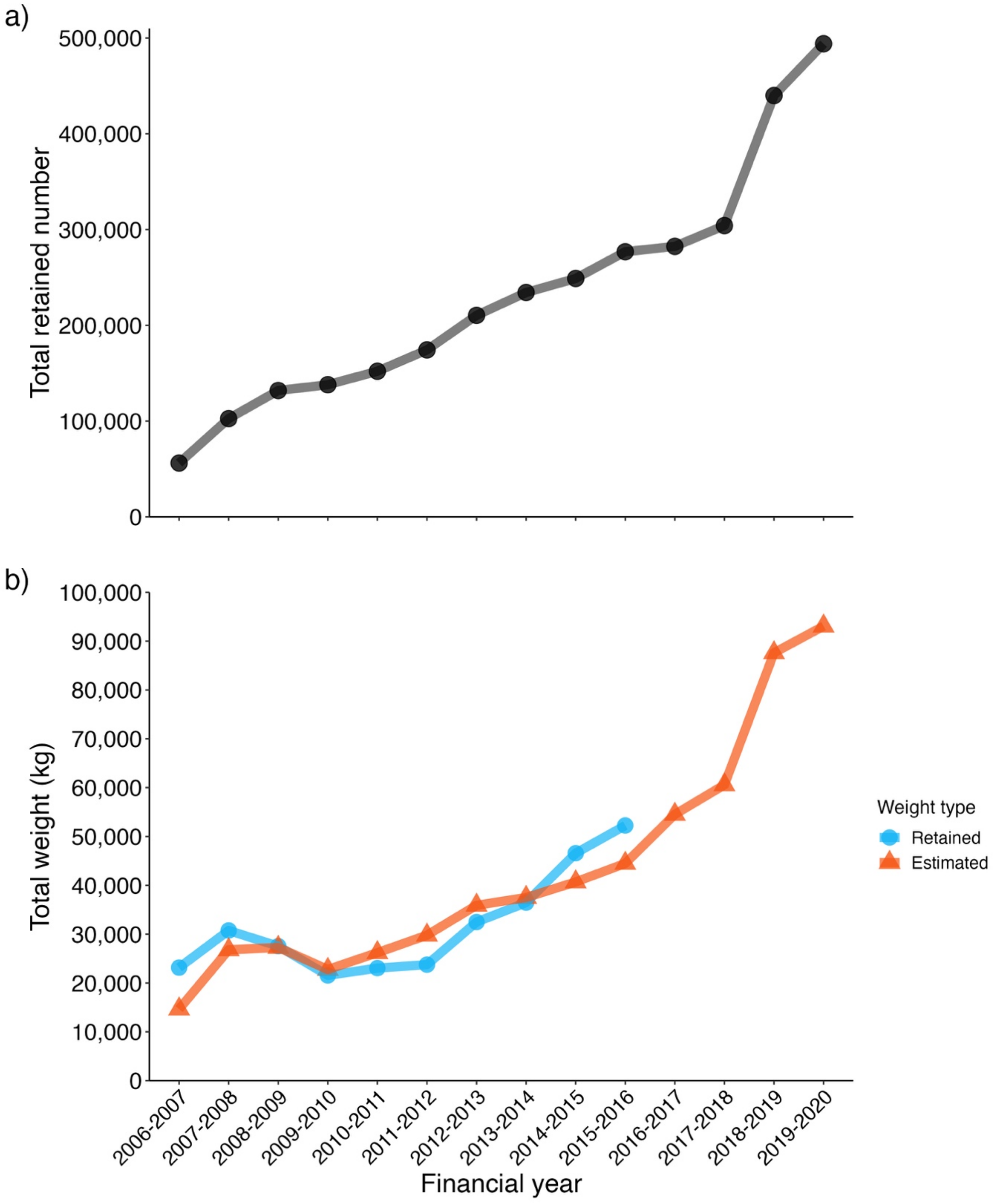

Figure 2.1. Annual reported harvest levels, based on a) no. of pieces retained and b) weight (kg) for live corals collected by the QCF for financial years 2006-07 to 2019-20. Estimated weight was calculated by multiplying the number of pieces by average weight per species or family to account for weight data from 2016-17 onwards. Harvest level data based on data

provided (DR35253iii) by Queensland Government Department of Agriculture and Fisheries, on May 17th, 2021.

Estimated weight increased 186% over the entire recording period, from 2006-07 at a value of just over 21,000 kg to around 93,000 kg in 2019-20 (2.1b). For retained weight, the average increase per year was 2,732 kg (Linear Regression; β estimate = 2732.0 ± 762.1 S.E.), while estimated weight over the same period was an estimated average increase per year of 31,777 kg per year (Linear Regression; β estimate = 3177.0 ± 377.4 S.E.). Comparison of regression coefficients suggest that the difference between retained and estimated weight was non-significant ($Z = 0.52$, $p = 0.3$), though sample size ($n = 10$) was low for this test type (recommended $n = 30$). The rate of increase for estimated weight between the years 2016-17 to 2019-20 was much steeper than in earlier years, at an average increase of 14,266 kg per year (Linear Regression; β estimate = 14266.0 ± 3011.0 S.E.).

2.3.2. Catch composition

Catch composition by family for scleractinian corals in (the most recent assessable year) 2019-20 was dominated by the SPS family Acroporidae (32.2%), followed by LPS Lobophylliadae (17.7%), LPS Merulinidae (13.2%), LPS Euphyllidae (11.1%), LPS Faviidae (8.2%), SPS Poritidae (5.4%), LPS Dendrophylliidae (3.2%), LPS Plerogyridae (2.0%), SPS Agariciidae (0.5), SPS Pocilloporidae (0.4%), and the lowest contributor LPS Oculinidae (0.3% ; Figure 2.2). Acroporidae contributed to catch composition just under twice (1.8 times) the amount compared to the next most significant contributor (Lobophyllidae). The first four families combined (Acroporidae, Lobophyllidae, Merulinidae, Euphyllidae) contributed 74.2% to catch composition for the 2019-20 financial year.

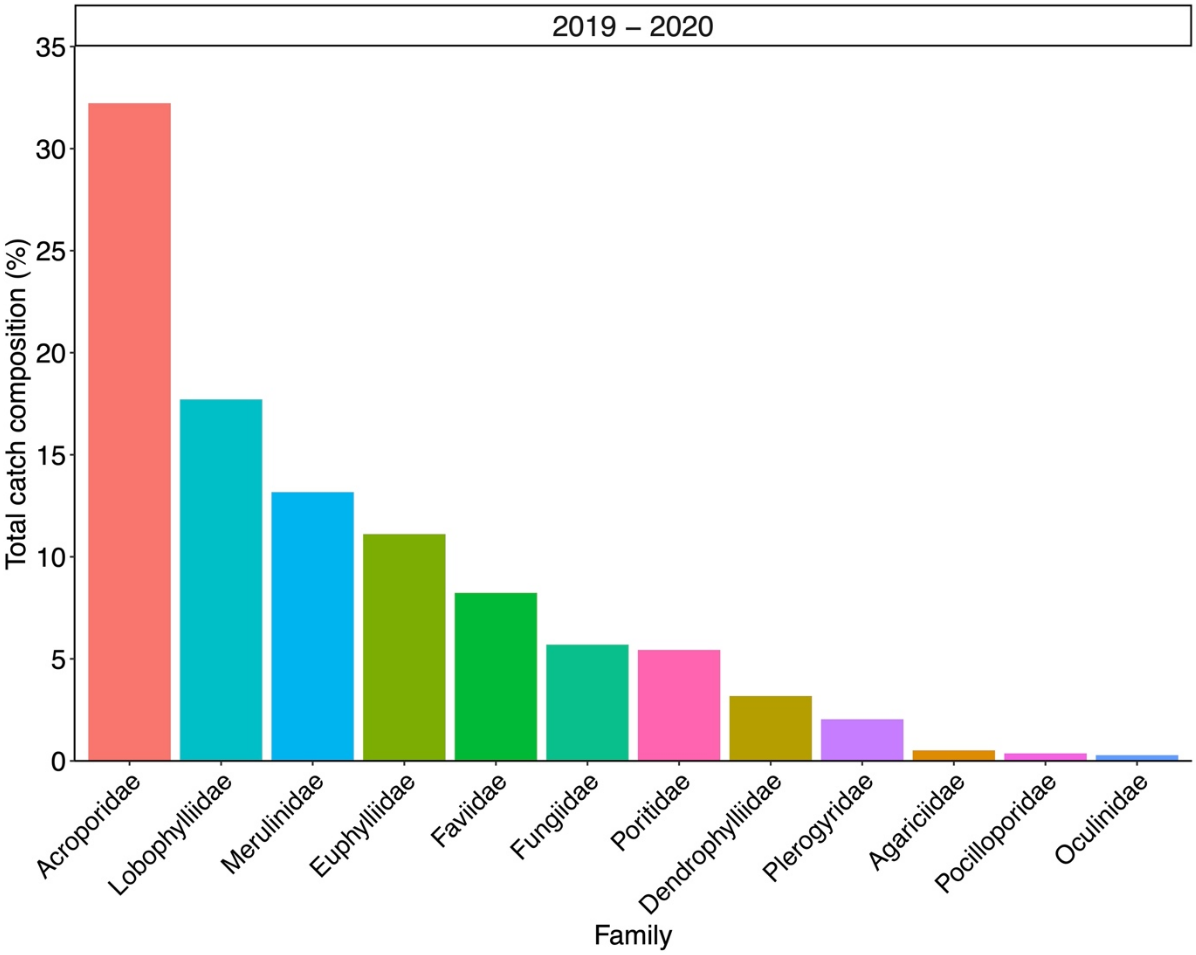

Figure 2.2. Catch composition by family (based on % no. of pieces) for hard corals (order Scleractinia) in the 2019-20 financial year. Harvest level provided by Queensland Government Department of Agriculture and Fisheries, May 17th 2021.

Catch composition for available taxa at the genus/species level was dominated by *Acropora* spp. corals, which alone accounted for 32.1% of total catch. This represents almost quadruple (3.82 times) the contribution of the next most significant contributor, *Homophyllia* cf. *australis* (8.4%). *H.* cf. *australis* is followed by the LPS species *Micromussa lordhowensis* (8.0%), *Trachyphyllia geoffroyi* (6.7%), *Goniopora* spp. (6.0%), *Fimbriaphyllia ancora*

(5.9%), *Catalaphyllia jardinei* (5.7%), *Euphyllia glabrescens* (5.1%), *Cycloseris cycloites* (3.7), *Montipora* spp. (2.9%), and *Duncanopsammia axifuga* (2.8%). These species combined accounted for 87.3% of total catch.

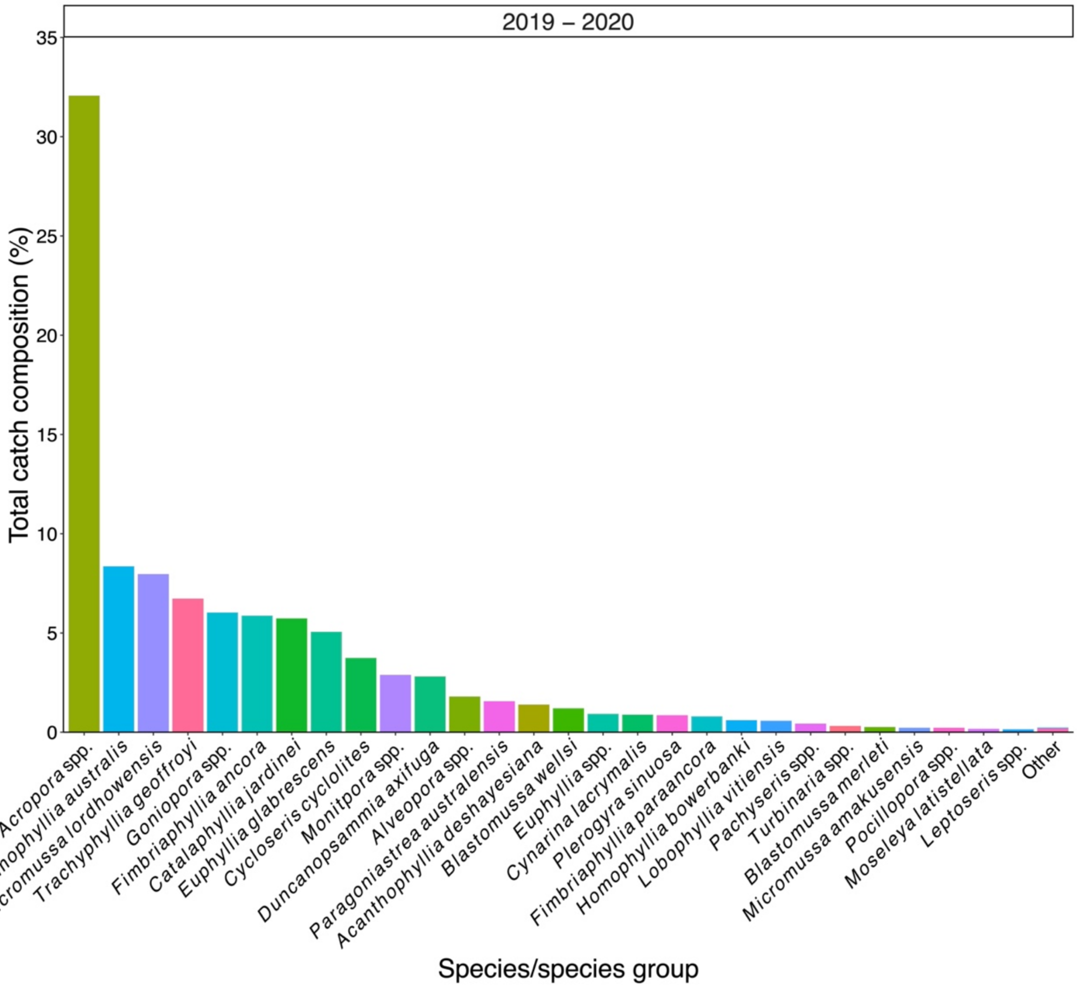

Figure 2.3. Catch composition by genera/species (based on % of pieces) for hard corals (order Scleractinia) in 2019-20. Harvest level data provided by Queensland Government Department of Agriculture and Fisheries, May 17th 2021.

2.3.3. Select species retained pieces and weight temporal trends

Species of interest each showed a distinct upwards trend identified by GAM (figure 2.4a). Long term trends in number retained pieces for SPS Acroporidae genera *Acropora* and *Montipora* were not recorded prior to 2016. Over the limited available time, a very sharp linear increase is obvious, with *Acropora* spp. jumping from over 82,000 retained pieces in 2016, to just below 139,000 pieces in financial year starting 2019, although peaking at around 144,000 pieces in 2018-19. This represents a 68% increase for *Acropora* spp. (Figure 2.4b), with an average increase per year of 18,865. The other SPS genus included in select temporal trend analysis, *Montipora* spp., has increased near exponentially (Figure 2.4a); from about 5,000 pieces in 2016-17 to just over 12,500 pieces in 2019-20 (148% increase; Figure 2.4b), averaging 2,487 piece per year increase.

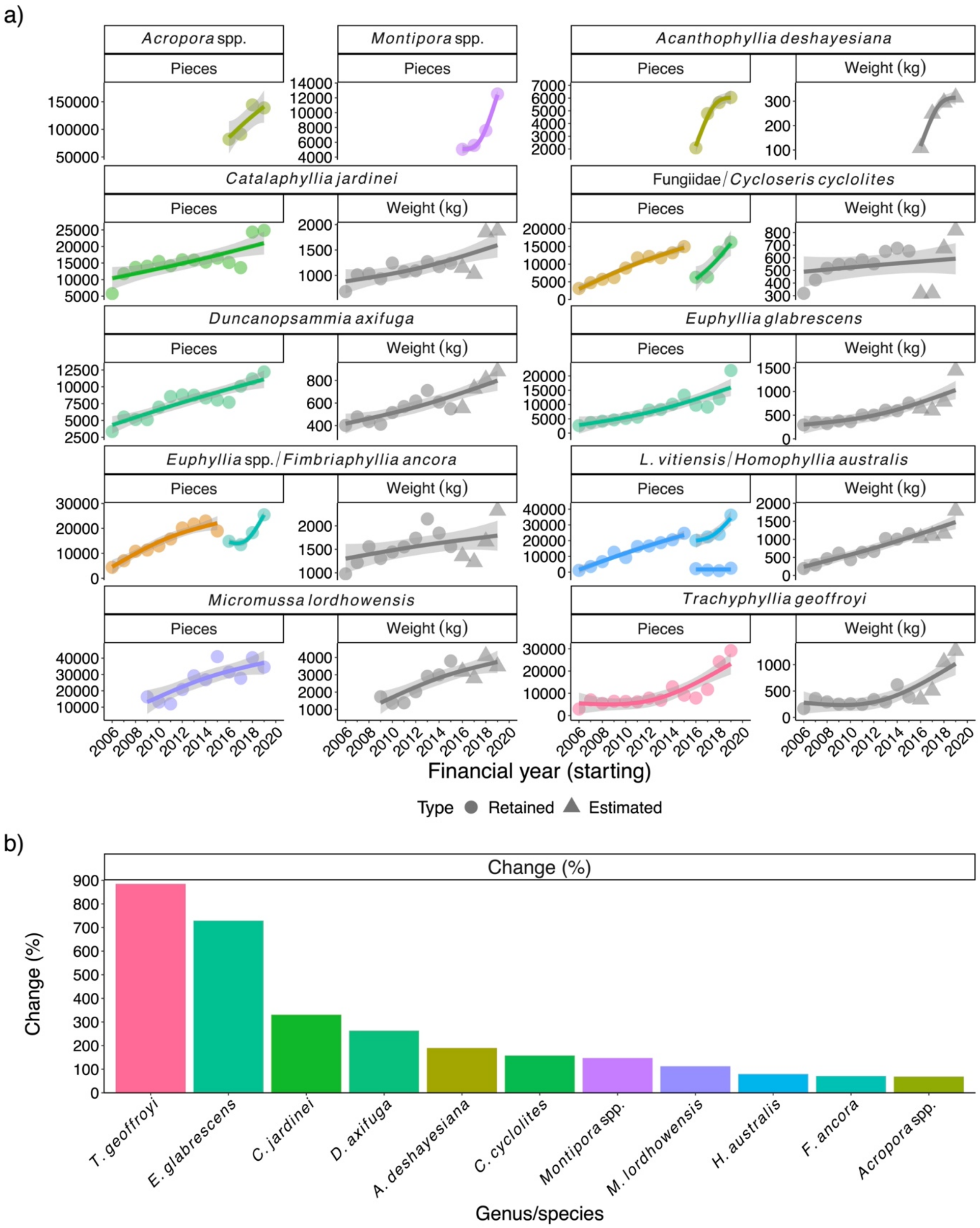

Figure 2.4. Results for select taxa of interest recorded to genera/species at a) annual reported harvest levels by unit (pieces, weight) and b) average change (%) calculated from initial retained pieces value, end retained pieces value and number of report category years within 2006-07 to 2019-20 financial years for single most recent report category only. Where reporting categories changed over recording timeframe, relevant groupings are displayed on

the same plot. GAMs (General Additive Model) were fitted to visually represent taxa current harvest trends, separating as necessary prior and contemporary recording categories. Coloured points represent total retained pieces recorded for that respective group in corresponding financial year, line represents GAM model, and shaded area represents 95% Confidence Band. Harvest level data provided by Queensland Department of Agriculture and Fisheries, on May 17th, 2021.

The LPS genus *Acanthophyllia deshayesiana*, which has only been recorded as a species level category since 2016-17 financial year, has also seen a steep increase with a logarithmic-like trend approaching 7,000 pieces (Figure 2.4a). This species saw a 190% increase over this period (Figure 2.4b), changing from a harvest level of around 2,000 pieces in 2016-17 to just over 6,000 by 2019-20 (average 1,323 pieces increase per year). Only estimated weight is available for this species, due to the creation of its species level category in 2016-17 financial year and the cessation of species level weight reporting for specialty corals. This species has now exceeded an annual harvest of 300 kg as of the 2019-20 financial year, increasing on average 69 $kg \cdot yr^{-1}$, from an initial value of 109 $kg \cdot yr^{-1}$ in the 2016-17 financial year (Figure 2.4a).

The LPS species *Catalaphyllia jardinei* displayed a steep increasing linear trend, although with some deviation above and below GAM model line and 95% confidence band (figure 2.4a). This species increased 331% (Figure 2.4b) from 5,700 pieces in 2006-07 to just under 25,000 pieces in 2019-20, which equates to an average increase of 1,469 pieces per year. *C. jardinei* had the third largest increase in retained pieces change percent out of selected species. Retained and estimated weight for this species reflected retained pieces. Annual harvest for this species has increased from 684 $kg \cdot yr^{-1}$ in 2006-07 to an estimated 1,887 $kg \cdot yr^{-1}$, averaging an increase of 92 $kg \cdot yr^{-1}$ (Figure 2.4a).

The LPS coral *Cycloseris cyclolites* was given its own category in the 2016-17 financial year, prior to which it is assumed to have been included in the Fungiidae family category. The temporal trend for this coral is a steep increase with a slight polynomial curve (Figure 2.4a). Harvest level for this species has increased 158% (Figure 2.4b) from 6,200 pieces in 2016-17 to around 16,000 pieces in 2019-20, averaging a 3317 piece per year increase. Estimated annual harvest in weight for this species exceeded 800 $kg \cdot yr^{-1}$ in 2019-20 financial year, increasing from an estimated initial annual harvest value of 316 $kg \cdot yr^{-1}$ in 2016-17 at a rate of 167 $kg \cdot yr^{-1}$ (Figure 2.4a).

The trend for the LPS coral *Duncanopsammia axifuga* was simplified to a steep increasing linear trend, though harvest points show deviation above and below the 95% model confidence band. This coral has experienced a 264% increase (Figure 2.4b) over the 14 recorded financial years, from over 3,300 pieces in the 2006-07 financial year to over 12,000 pieces in 2019-20, with an average increase of 680 pieces per year. *D. axifuga* has increased from a retained initial annual harvest of 402 $kg \cdot yr^{-1}$ in 2006-07 to an estimated 880 $kg \cdot yr^{-1}$ in 2019-20, with an annual increase of 37 $kg \cdot yr^{-1}$.

Euphyllia glabrescens, had a steep, increasing, slightly parabolic trend over the 14 financial year period of recorded data. This LPS species increased 729% (Figure 2.4b), from around 2,600 pieces in 2006-07 to just under 22,000 pieces in 2019-20 (Figure 2.4a). This was the second largest percent increase out of selected species (Figure 2.4b). *D. axifuga* has increased from an initial retained annual harvest level of 295 $kg \cdot yr^{-1}$ in 2006-07, to an estimated 1,445 $kg \cdot yr^{-1}$ in 2019-20, averaging an 88 $kg \cdot yr^{-1}$ increase.

Fimbriaphyllia ancora (prev. *Euphyllia ancora*; Luzon et al., 2017) received its own species level recording category in 2016-17, prior to which it was presumably included in the *Euphyllia* spp. genus category. Harvest for this species over time initially dipped in the year following category creation (2017-18), and then increased sharply. *F. ancora* increased 68% (Figure 2.4b) over the 4-year recording period, with an initial harvest level just shy of 15,000 pieces in 2016-17 to over 25,000 pieces in 2019-20, with an average increase per year of 2,949 pieces (Figure 2.4a). Estimated weight (kg) for *F. ancora* has increased from 196 kg·yr^{-1} in 2016-17 to 319 kg·yr^{-1} in 2019-20, increasing 41 kg·yr^{-1}.

Homophyllia cf. *australis* is an iconic Australian LPS coral that has seen a near exponential harvest level increase since the creation of its species-specific report category in 2016-17 (Figure 2.4a). Prior to the creation of this reporting category, this coral was presumably recorded as *Lobophyllia vitensis*. As indicated by the harvest level of *L. vitensis* post generation of the category for *H.* cf. *australis*; *H.* cf. *australis* was most likely the major contributor to this category. *H.* cf. *australis* has seen a 79% increase (Figure 2.4b) from around 20,000 pieces in 2016-17 to over 36,000 pieces in 2019-20, with an average rate of increase of 5,363 pieces per year. This species has seen an average increase of 253 kg·yr^{-1}, from an estimated 1,037 kg·yr^{-1} to an estimated 1797 kg·yr^{-1}.

Micromussa cf. *lordhowensis* has been recorded in a species-specific category since the 2009-10 financial year. This species has seen a steep increase, with a slight parabolic curve and some notable deviation above and below model 95% confidence band (Figure 2.4a). This species has experienced a 112.4% increase (Figure 2.4b) over the 11 financial year period, from over 16,000 pieces in 2010-11 to around 34,500 pieces in 2019-20, increasing at an average rate of 1,826.0 pieces per year. This species has seen a 177 kg·yr^{-1} increase in harvest level, going from 1,731 kg·yr^{-1} in 2009-10 to an estimated 3,495 kg·yr^{-1} in 2019-20.

Trachyphyllia geoffroyi is another major target LPS species. The temporal trend visualised by GAM model line is a parabolic increase marked by a significant and sudden increase in the final two financial years (2018-19 and 2019-20). This species experienced the most significant percent change increase, at 884.8% over the 14-year period (Figure 2.4b), from just under 3,000 pieces in 2006-07 to just over 29,000 pieces in 2019-20, averaging an increase of 2,015 per financial year (Figure 2.4a). The retained weight for this species was 168 $kg{\cdot}yr^{-1}$ in 2006-07, increasing up to an estimated 1,265 $kg{\cdot}yr^{-1}$in 2019-20, at an average 85 $kg{\cdot}yr^{-1}$.

2.4. Discussion

The Queensland Coral Fishery is the largest of these operations in Australia in both geographic extent and total annual commercial catch (TACC). Since the inception of this fishery in the 1980s, harvest composition has continued to shift to reflect market demand, from skeleton curios, to live rock, onto LPS corals; and now into the era of an SPS *Acropora* dominated fishery (at least by single, available report category; Pratchett, 2021a; Harriott, 2003). Since the establishment of contemporary management strategies in the 2006-07 financial year, including the implementation of a roving license approach, total catch in pieces has increased over 750% compared to total catch in the most recently assessed 2019-20 financial year; averaging a 21,675 piece per year increase in annual harvest. The increase in total weight (kg), while still substantial, did not match the increase in collected pieces. This indicates that the increase in number of pieces has corresponded with a reduction in the size (and so, weight) of retained coral pieces. Despite this dramatic increase, annual harvest levels are still below the TACC for SPS (i.e., ‘other coral’;140 t) and LPS (i.e., ‘specialty coral’; 60 t).

2.4.1. Significance of Acroporidae

Acroporidae was the most significant contributor to catch composition at the family level, a trend likely driven by the considerable volume in number of pieces harvested for the genus

Acropora, which eclipsed other categories at the genus/species level accounting for over 32% of catch composition among available coral species categories. According to CITES exporter reports, approximately 361,121 pieces of Acroporidae were in circulation globally in the year 2010 (Wood et al., 2012). Roughly 9 years later, the total annual harvest for *Acropora* spp. in Australia alone is 144,367 pieces in the most recently assessed year (2019-20), which represents 40% of total historic (2010) world trade. Acroporidae have always formed a major component of 'other coral' harvest in Australia (i.e., falling under the 140 t quota split), though the harvest methodology and target taxa has changed considerably in the last two decades. From before the implementation of the contemporary roving harvest strategy (i.e., 2006-07 financial year), fishers began to move away from the collection of larger, heavier intact coral skeletons for curios (i.e., that may or may not have been dead prior to collection; Harriott, 2001) in favour of smaller live pieces due to the popularity of *Acropora* in 'mini-reef' style home aquaria following improvements in technology (Dee et al., 2014; Rhyne et al., 2012; Wood et al., 2012). This shift meant that the majority of the 'other corals' quota, initially intended for the collection of coral skeleton curios, was now comprised of mostly live coral pieces, presumably also of greater phenotypic diversity. While this represents a significant change from the intended catch composition for the assigned 140 t 'other coral' quota, the potential implications of this change for harvest sustainability have not been explicitly considered. More recently, contemporary export bans in the major SPS (esp. *Acropora* spp.) exporting nation Indonesia from 2018 to 2020 (Chalias, 2019) are also likely to have contributed to the increased demand for this genus, perhaps as indicated by notable surge in demand shown in annual harvest data post-ban. The exponentiating harvest levels of SPS taxa is one signifier of the need for a shift in management attitude and approach for these corals considering mounting environmental pressures.

As the largest single group contributing to catch composition at the family and species level, further consideration to harvest management strategy is required to account for the complexity of managing *Acropora* and other major SPS Acroporidae genera. *Acropora* is a major reef building taxa, accounting for 30% of hermatypic coral species (Madin et al., 2016). When considering harvest of *Acropora* at a genus level, harvestable standing biomass in the Great Barrier Reef Marine Park must greatly eclipse the 140 t harvest limit set from 2006-07 to 2019-20 (Chapter 3, Pacey et al., 2022). However, while this fast growing genus is highly prevalent (Pratchett et al., 2015), *Acropora* colony longevity (i.e., population level dynamics) and size (i.e., individual level dynamics) is tightly constrained by stochastic disturbance events such as cyclones and tropical storms, coral predator outbreaks, as well as mass thermal coral bleaching events that can result in largescale whole or partial mortality (Pratchett et al., 2013; Wakeford et al., 2008). At least some of these major disturbance events (e.g., cyclones and severe storms, mass coral bleaching) are likely to increase in intensity and/or frequency due to anthropogenic climate change (Cheal et al., 2017; Hughes et al., 2017a; Kang and Elsner, 2015). Therefore, the cumulative and catalytic impact of unmitigated increases in coral harvest, when simultaneously co-occurring with more frequent disturbances of greater intensity, should be accounted for in harvest management strategy with taxa specific limits.

2.4.2. Implementation of genus and species level harvest limits

For the first time in the QCF's history, genus and species level harvest limits were implemented for 11 taxa as a result of considerable and unmitigated growth in total harvest (Pratchett, 2021a). This included 2 SPS genera *Acropora* and *Montipora* (Family Acroporidae), and 9 LPS species *Acanthophyllia deshayesiana* (F. Lobophyllidae), *Catalaphyllia jardinei* (F. Merulinidae), *Cycloseris cyclolites* (F. Fungiidae), *Duncanopsammia axifuga* (F. Dendrophyllidae), *Euphyllia glabrescens* (F. Euphyllidae), *Fimbriaphyllia ancora* (F. Euphyllidae), *Homophyllia* cf. *australis* (F. Lobophyllidae), *Micromussa* cf. *lordhowensis* (F. Lobophyllidae), *Trachyphyllia geoffroyi* (F. Merulinidae). While the QCF harvest strategy 2021-2026 (DAF, 2021) allowed for continued increase in the harvest level of all coral taxa with the exception of *Homophyllia* cf. *australis*, as recommended during the most recent reassessment of approved wildlife trade operation (Commonwealth of Australia, 2021; Pratchett, 2021a), contrary action was taken to constrain catch quota to 80% of average harvest over the contemporary harvest period (2016-17 to 2019-20) for species of moderate concern (*Acropora* spp., *Micromussa* cf. *lordhowensis*, *Homophyllia* cf. *australis*, *Trachyphyllia geoffroyi*, *Acanthophyllia deshayesiana*; Roelofs and Albury, 2018), and to 100% of reference period for taxa of low concern (*Montipora* spp., *Catalaphyllia jardinei*, *Cycloseris cyclolites*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Fimbriaphyllia ancora*; Roelofs and Albury, 2018). Using only report category data (i.e., excluding historic groups splits), 7 of these taxa have undergone an increase of total retained pieces exceeding 100% from initial reporting year to latest assessment year (2019-20; Figure 2.4b). Three of these taxa (*T. geoffroyi, E. glabrescens, C. jardinei*) have exceeded a 300% increase. Perhaps most notably, *T. geoffroyi* has seen an increase of just under 900% in the 14 financial year reporting period. This solitary inter-reefal LPS coral is patchily distributed (commonly found around continental island shelves), and is thought to have slow and highly stochastic (i.e., highly environmental condition dependent) growth (Pratchett et al., 2020a). As these characteristics make this species highly vulnerable to localised depletion, the seemingly unmitigated increase in harvest for this species over the examined period has the potential to pose a serious risk to local *T. geoffroyi* populations; particularly considering the lack of quantitative data to inform estimates of abundance and distribution. Interestingly, the top 3 taxa (*T. geoffroyi, E. glabrescens, C. jardinei*) that have experienced the greatest percent increase all have species level reporting categories spanning the entire sample period. The percent increase for other select species may in fact be more dramatic if pre-split historic data is also incorporated. For example, an increase of 3,074% has occurred for the heavily targeted solitary inter-reefal LPS coral *H.* cf. *australis* if prior recording category of *L. vitensis* is used to inform harvest prior to 2016-17 (following updated taxonomic information; Arrigoni et al., 2016). In combination with a variety of (increasingly rare) striking colour morphs, due to the limited distribution of *H.* cf. *australis* (i.e., Australian waters), this coral is highly prized due to the perceived 'rarity' within the ornamental marine aquarium trade (Rhyne et al., 2012). While species specific harvest limits have necessarily been adopted in response to the heightened risk of deleterious impacts to populations associated with unmitigated harvest increase, a crucial need still exists for ecologically relevant and informed justification for harvest limits.

2.4.3. Implications for harvest

A set harvest limit (or quota) is only (reliably) effective if it informed by the biological, ecological, and economic information necessary to justify the prescribed level of exploitation as sustainable (Beddington et al., 2007; Rhyne et al., 2009). Contrary to historic perception that Australian coral fisheries are sustainably managed, there has been limited quantitative data, or even robust mathematical justification for its proposed harvest limits. The work of Harriott (2001) has often been used as evidence to suggest that Australia's coral harvest fisheries are well managed (e.g., see Dee et al., 2014; Rhyne et al., 2014; Wood et al., 2012). The 'less than 1% of available coral could be harvested' justification for sustainability was postulated prior to the adoption of a roving license approach, when the area open to harvest existed in 50 small spatially specified zones averaging 25,000 m^2, each tied to a single harvest license (Harriott, 2001). This estimation did not account for the distribution of contiguous reef or target LPS taxa, considering only total park area open to harvest, which was considerably less compared to the present. This may have set a dangerous precedent, as there is no quantitative or mathematical justification provided for the contemporary harvest limit of 200 t (60 t specialty coral, 140 t other coral), other than the sheer geographic scale of area open to harvest and the abundance of target taxa assumed to exist across such a vast area. Observed catch-based assessment, even with defined management actions in addition to input and output controls, can provide a false sense of security (Ovando et al., 2022; Walsh et al., 2018). If there is no quantitative evidence to support that past reference harvest levels are sustainable, then misleading exploitation baselines may be established (e.g., see Garrabou et al., 2017). As implied, in such circumstances fisheries operating within prescribed harvest limits are still at significant risk of over-exploitation even when controlling the rate of harvest increase. Simply, if the ability to continue or increase the rate and/or quantity of harvest each season is used as evidence for sustainable harvest, unmitigated harvest increase can be considered as sustainable until inarguable signs of overexploitation (i.e., ranging from diminishing effort returns to total fishery collapse) are apparent (Rosenberg, 2003).

Furthermore, even the implied ‘sheer scale’ justification of contemporary harvest quota is likely irrelevant when considering the possibility of local depletion for many heavily targeted LPS species due to their distribution and the spatial concentration of harvest. The maximum percent of harvest occurring within a 6x6 nautical mile (nm) block over the four most recently assessed financial years (2016-17 to 2019-20) exceeded 10% for all LPS taxa of interest examined in this study (see Table 4.3, Pratchett, 2021a). For 5 select species (*M.* cf. *lordhowensis*, *C. cyclolites, A. deshayesiana, E. glabrescens*, and *T. geoffroyi*) all exceeded 15% of total annual harvest within a single 6x6 nm block, and 3 met or exceeded 25% (*M.* cf. *lordhowensis*, *C. cyclolites, A. deshayesiana*). The maximum percent occurring within a 6x6 nm block averaged 33.1 (± 4.5 S.E.) for *M. lordhowensis*, with one year (2017-18) even exceeding a maximum percentage of 41.8 % of all harvest (in pieces) occurring within a 6x6 nm block. As the total pieces harvested that year for *M. lordhowensis* was 27,702 pieces, this represents an approximate 11,579 pieces from an area spanning roughly 11.2 km^2. When 41.8% (~11,579) of harvest occurs in an area representing of 3.2×10^{-3} % of the total park area (344,400 km^2), the risk of local depletion must be considered astoundingly high in the absence of quantitative ecological data; particularly as a logical conclusion to be drawn from this concentration of harvest is that though widely distributed according to Veron et al. (2021), profitably concentrated aggregations of this species exist only in very limited spatial extents. Compounding this issue in cases of concentrated harvest, the roving license approach (particularly when coordinate data is not available for analytic use) seriously impedes the capacity to properly quantify risk and impact for these species when harvest is concentrated, but the management area is vast. Species specific monitoring is needed for key target species (see Pratchett, 2021a), particularly if harvest limits are to be released from current restrictions.

2.4.4. Considerations for improving management

Quantitative field data and biological parameters are required to verify that contemporary harvest levels do not pose a risk of local depletion to species at a harvest-relevant geographic scale. Investment in the collection of *in situ* species specific monitoring data is not just necessary from a scientific perspective, it is in the interest of managers hoping to repair the crumbling global reputation of sustainable Australian coral harvest, as well as fishers currently faced with reduced harvest potential for a number of high value species. Species specific in situ monitoring will be required to establish sustainable baselines and allow for the retirement of current reduced species-specific quota. While the need to establish defensible, quantitative justification for harvest limits that include appropriate baselines may be obvious, the methods to accomplish this are far less so. In the case of the QCF, it is difficult to establish a 'baseline' without any capacity to bridge the gap between harvest levels and ecological impact. Translating harvest units (weight quota) into ecologically relevant impacts presents a significant challenge given the range of life-strategies employed by scleractinian corals (Chapter 3 & 4; Longenecker et al., 2015; Pacey et al., 2022, 2023). Compounding this, is the immense harvest area (permitted zones of the 344,400 km^2 Great Barrier Reef Marine Park), lack of biological information for LPS species, and need for the development of ecosystem based management capacity for stochastic disturbance prone SPS species such as *Acropora* (Pratchett, 2021a; Pratchett et al., 2020a). Importantly, although species level (and harvest relevant) reporting categories are crucial to sustainable management, this is a particularly lofty goal for some taxa; even assuming funds and survey effort are not limiting factors.

For example, *Acropora* is a highly speciose genus currently in the midst of major taxonomic revision (Ball et al., 2021; Bridge et al., 2023). Harvest of *Acropora* corals is highly selective, focusing on distinct phenotypes (e.g., *Acropora* cf. *microclados* ‘strawberry shortcake’). Some currently recognised species such as *Acropora* cf. *microclados* are considered to be vulnerable globally (IUCN, 2008). For this reason, assessment at the genus level over the expansive area of the QCF may be particularly deceptive, particularly in light of the reality of spatially concentrated (and economically viable) harvest as seen for LPS species (Pratchett, 2021a). Furthermore, without an established species level taxonomy, it is difficult to speculate not just on the abundance of particular species, but also the inheritance of particular phenotypic traits. Distinct, desirable phenotypes of a species may in fact represent an entirely different species altogether (Bridge et al., 2023). Alternatively, even if desirable phenotypes are a result of expression within a defined species, the inheritance of these traits could create negative selection pressure for this trait in the population due to fishing (Allendorf and Hard, 2009). Phenotypes could also be geographically restricted not just by genetics, but also by environment and/or location, as gene expression has been linked to environmental variables in *Acropora* and other scleractinian colonial branching corals (Granados-Cifuentes et al., 2013; Hackerott et al., 2023; Padilla-Gamiño et al., 2012; Todd, 2008). Management scale remains a persistent challenge in this fishery for both SPS and LPS corals. To properly account for harvest impacts, the abundance of phenotypes that are commonly harvested, as well as the distribution of harvesting efforts must be quantified.

2.4.5. Conclusions

In light of the significant increase in reported harvest, as well as failure to initiate defined harvest trigger thresholds (see DAF, 2016), temporary harvest limits (Pratchett, 2021a) have been established for taxa in an interim capacity, with the intention of updating these limits on a species-specific basis when sufficient ecological basis for harvest can be established (Commonwealth of Australia, 2021). These limits have been set according to perceived threat to harvested taxa based on an ERA (Ecological Risk Assessment) incorporating catch as well as any biological and ecological data. These harvest limits have been set using a multiplier encapsulating perceived risk combined with a reference harvest level (set for the period between 2016/17 to 2018/19). *Acropora* spp., *M. lordhowensis*, *H.* cf. *australis*, *T. geoffroyi*, *A. deshayesiana* were all considered to be of ‘moderate’ concern, and so a multiplier of 0.8 was applied (Pratchett, 2021a). The taxa *C. jardinei*, *F. ancora*, *E. glabrescens*, *D. axifuga*, *C. cycloites*, *Montipora* spp. received a multiplier of 1.0 being of ‘low’ concern, and all other taxa were constrained to a 150% increase from the same reference harvest level (Pratchett, 2021a). As the newly implemented harvest limits mark a change in attitude towards the management of this fishery, the need to improve quantitative data input capacity and resolution remains a persistent issue (Harriott, 2001). Coral harvest fisheries in Australia face internal (Commonwealth of Australia, 2021) and/or external (European Commission, 2023; Gay, 2022) market access restriction as a result of reactive (as opposed to proactive) management decision making. However, catch based management in data-limited fisheries necessarily limits the capacity to plan management actions prior to harvest reports (Nielsen et al., 2018; Walsh et al., 2018). With the required management goal (i.e., implementation of quantitative inputs) clear, the question of what these inputs should be, the methodology required to implement these input sources, and how they can be best utilised remains largely indeterminate. Firstly, focused and strategic monitoring efforts are required to establish estimates of the current standing biomass of targeted corals in heavily fished areas. Additionally, adjacent unfished zones should be used where possible/suitable to establish a reference baseline. Methodology to estimate the available biomass of corals using size weight relationships was established prior to (Longenecker et al., 2015), and especially following the previous wildlife trade operation assessment (Chapter 3 & 4; Pacey et al., 2023, 2022). The use of video transect surveys provide a means of estimating biomass along the length of a transect, which can be used to characterise a defined area with appropriately placed repeated sampling. Sampling of this nature can be used to link harvest in quota relevant units (i.e., weight) to spatially bounded harvest data (Chapter 3 & 4; Pacey et al., 2022, 2023; Pratchett et al., 2020a). Furthermore, it is important that both weight and retained piece units are recorded at the lowest taxonomic resolution possible, as has been instituted post-assessment (Pratchett, 2021a), as this provides a means with which to assess historic trends against current trends, as well as generate weight per piece estimates. Secondly, further development of fishery relevant biological information is required for both SPS and LPS species, though the priority of knowledge gap bridging exercises differs between the groups. Key biological and ecological parameters for heavily targeted LPS species remain largely undefined. *In situ* growth, recruitment, reproduction, as well as factors underlying spatial distribution would be of particular relevance to understanding stock dynamics. For SPS species such as the dominant group *Acropora*, the potential for localised depletion cannot be properly assessed until a working taxonomy is established that can allow quantification of (at least a quasi-) species-specific standing biomass for heavily targeted species/phenotypes. Therefore, developing a working taxonomic grouping relevant to the fishery should be the top priority for managing *Acropora* corals. Thirdly, consideration of a new quantitative stock assessment framework is required. Ideally, a quantitative framework will be based on an underlying stock assessment model (Gulland, 1983; Nielsen et al., 2018). As the life strategies of SPS and LPS solitary taxa represent two diverging extremes, establishing a model for each type using a prominent reference species may allow development of modelling strategies that could be tailored to species falling between these in terms of growth, coloniality, and harvest vs environmental risk. Failure to implement these actions is likely to lead to continued trade bans of increasing duration, severity, and taxonomic bredth, which due to the international market access required for coral harvest fisheries to remain economically profitable, could eventually result in the economic strangulation of this industry. Therefore, although quantifiable and defensible management will require considerable further investment; through strategic investment and capacity building, Australia is presented with a unique opportunity to establish the precedent for sustainably managed coral harvest fisheries globally.

Chapter 3

Size-weight relationships for estimating harvestable biomass of *Acropora* corals on Australia's Great Barrier Reef[1]

3.1. Introduction

Coral reefs are among the most diverse and productive marine habitats, but are highly vulnerable to the escalation and intensification of anthropogenic pressures (Albert et al., 2015; Grafeld et al., 2017; Moberg and Folke, 1999; Spalding et al., 2017), and particularly increasing effects of anthropogenic climate change (Bruno and Valdivia, 2016; Hughes et al., 2017a), as well as continued eutrophication, sedimentation, and pollution resulting from coastal development and modification (Fabricius, 2005; Wear, 2016). Even intensively managed and extensive systems such as Australia's Great Barrier Reef (GBR) have been subject to pervasive and continuing degradation; De'ath et al. (2012) documented a 50.7% decline in average coral cover from 1985 to 2012. More recently, Mellin et al. (2019) predicted that coral cover had declined on over 90% of reefs within the GBR between 1996 and 2017, with an annual mean coral cover loss of -0.67% steepening to -1.92% from 2009 to 2016. Recurrent episodes of climate-induced coral bleaching events are further exacerbating coral loss and reef degradation (Hughes et al., 2018a, 2018b), placing vulnerable coral reef species, ecosystem function and integrity at risk (e.g., McWilliam et al., 2020).

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Widespread and accelerating degradation of coral reef ecosystems place increasing environmental, public, and political pressure on extractive or potentially destructive activities (Burke et al., 2011; Hughes, 1994; Negri et al., 2002), especially the direct harvest of reef-building corals. Ecological concerns regarding the ongoing wild harvest of hard corals relate to the potential for localised depletion or compromised resilience of heavily targeted or highly vulnerable coral species, especially given the potentially cumulative and catalytic nature of these impacts when combined with other chronic and acute drivers of coral loss (Dee et al., 2014; Harriott, 2003; Jones, 2011; Wood et al., 2012). Increasingly, a portion of traded corals (especially Acroporidae) are produced or grown-out through aquaculture and mariculture operations (Barton et al., 2017; Craggs et al., 2017; Nietzer, 2016), although the vast majority are still harvested from the wild (Rhyne et al., 2012; Wood et al., 2012). The wild harvest and global trade in hard corals faces many of the challenges typical of other fisheries (Beddington et al., 2007), especially limited fishery-independent data necessary to establish the sustainability of escalating harvest levels (Bruckner and Borneman, 2006; Petrossian et al., 2020; Sosnowski et al., 2020). This is despite the fact that all hard corals are listed under Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) which should ensure that harvest levels and practices do not unduly jeopardise coral stocks and species (Wood et al., 2012).

Australian coral fisheries are small-scale and highly selective (i.e., hand-collection), and are often regarded as well-managed (Harriott, 2003). However, the ecological impact and sustainability of these fisheries is difficult to reconcile. The Queensland Coral Fishery (QCF) operates along the length of the Great Barrier Reef Marine Park (GBRMP), which spans an area of ~344,400 km^2 and contains an estimated 20,679 km^2 of shallow reef area (Near Sea Surface coral reef, <20 m depth) (Harris et al., 2013). Shallow reefs account for ~6% of the total area, and just over 33% of the total GBRMP area is closed to fishing (Fernandes et al., 2005). The QCF has the largest catch quota of Australia's coral fisheries with a Total Allowable Commercial Catch (TACC) of 200 t per annum. The QCF's TACC quota is split into 30% (60 t) 'specialty coral' and 70% (140 t) 'other coral' categories (DAF, 2020). Historically, fast-growing reef-building corals (Acroporidae and Pocilloporidae) were collected for export as curios and were therefore pooled with 'live rock' (i.e., dead coral rubble colonised by coralline algae and other non-CITES listed biota) into 'other coral'. There has however, been a recent resurgence in trade of *Acropora* as live aquarium specimens (Rhyne et al., 2012; Sosnowski et al., 2020), and a systematic decline in the trade of curios and live-rock (Wood et al., 2012). In 2010, > 93,000 pieces of live *Acropora* were imported into the US from wild harvest fisheries (Wood et al., 2012). *Acropora* coral forms a major component of the QCF harvest, representing 27% (> 88,000 pieces) of total coral harvested in 2016-2017. Based on current quota arrangements for the QCF, > 140 t of live *Acropora* could be theoretically harvested from the GBR every year (DAF, 2018, 2020). Current reported catch for Acropora spp. corals has continued to increased steeply in the QCF between the 2016/17 and 2019/20 finacial years (Chapter 2; Pratchett, 2021a).

Growing pressure to provide ecologically sustainable, low impact, and origin traceable products has resulted in increased social, economic, and ecological pressure on coral collectors and suppliers (Pratchett et al., 2020a). At currently reported harvest levels and limits, coral harvesting represents a very moderate source of coral loss in Australia, especially compared to major disturbances such as cyclones, crown-of-thorns starfish outbreaks, or mass coral bleaching (De'ath et al., 2012; Hughes et al., 2018a; Mellin et al., 2019; Pratchett, 2010). However, increasing incidence of major disturbances and corresponding declines in abundance of hard corals also increases the imperative of fisheries to demonstrate that harvest levels and practices are having negligible or acceptable impacts on coral stocks and species. Failure to reconcile coral fishery impacts, especially in light of increasing anthropogenic and environmental pressures on reef ecosystems, could jeopardise international demand and export approvals for reef-building corals harvested across Australia.

A number of approaches have been implemented previously to quantify and monitor environmental impacts of coral fisheries. Bruckner and Borneman (2006) used species density data matched with habitat occurrence and availability data to assess the sustainability of catch quotas for a range of taxa targeted in Indonesian coral fisheries. This approach was effective for Indonesian fisheries, which at the time were mainly targeting free-living, often solitary coral taxa that have relatively uniform growth forms, and are collected as whole colonies (i.e., quantified in terms of individual count). For the harvest of *Acropora*, however, as well as many other colonial coral species that may vary greatly in size and are often sub-sampled (fragmented) during harvesting, the density of corals (no. of colonies per unit area) is largely irrelevant. Rather, estimates of harvestable or standing biomass (combined weight of all colonies within a prescribed area) are needed to effectively assess the suitability of coral quotas and potential ecological effects of harvesting (Ross, 1984). The individual weight and combined biomass of corals is difficult to measure directly, but may be inferred based on known relationships between size and weight (e.g., Longenecker et al., 2015). Longenecker et al. (2015) estimated harvestable biomass for *Acropora* colonies to establish sustainable harvest limits of *Acropora* coral for lime production in Kamiali Wildlife Management Area, Papua New Guinea. However, size-weight relationships were determined for a relatively limited ($n = 60$) sample of colonies and considered only one (corymbose) growth form, even though a diversity of different *Acropora* corals were commonly harvested (Longenecker et al., 2015). Similarly, *Acropora* corals harvested by the QCF display a wide range of growth forms, though specific preferences for particular species or growth forms of *Acropora* are yet to be properly established.

As a first step towards ascertaining ecologically sustainable harvest limits for *Acropora* corals on Australia's Great Barrier Reef, this study seeks to establish size-weight relationships for relevant growth forms of *Acropora* harvested by the QCF. These size-weight relationships are then used to estimate the harvestable biomass of phenotypically diverse *Acropora* corals at select sampling locations encompassing a range of habitats and regions along the GBR. As ecological data relevant to regions of operation for the QCF will be required for biomass per unit area estimation, there is further opportunity to note any differences in available compared to collected *Acropora* growth forms and biomass. Establishing these size-weight relationships will provide quantifiable ecological context to QCF operations, and may also be a broadly useful tool for operators, managers, monitors, and researchers (Wolfe et al., 2020).

3.2. Materials and methods

3.2.1. Data collection

Size-weight relationships for *Acropora* were determined based on 3,826 samples provided by licenced coral collectors from the Queensland Coral Fishery (QCF), located in Cairns and Mackay, Queensland, Australia (Figure 3.1). The average diameter (based on maximum and perpendicular diameter), weight, species name, and type (fragment, colony, colony with base) were recorded for all individual *Acropora* colonies or fragments, with all sampling conducted between August and November, 2019. Diameters were recorded to the nearest centimetre (cm), and weights were recorded to the nearest gram (g). Corals were then categorised into one of 6 growth forms: bottlebrush, caespitose, corymbose, digitate, staghorn, and table, following Veron (2000). The 'type' category was used to capture whether the coral piece was at the time of measurement a fragment (i.e., piece) of a colony or an entire colony, and whether the colony had a 'base' (i.e., a large piece of substrate or non-living coral skeleton located underneath the colony) attached.

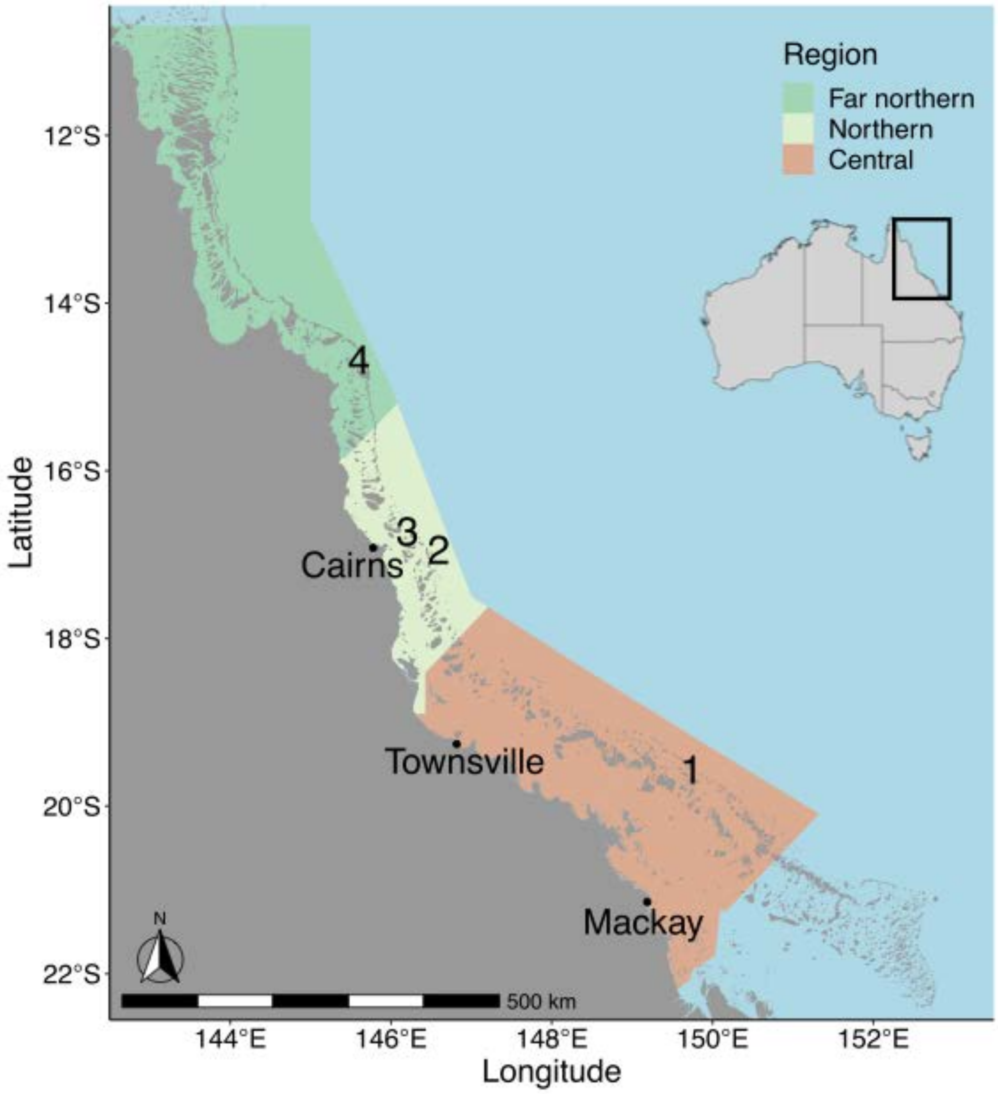

Figure 3.1. Survey regions ('Central', 'Northern', 'Far northern') relative to the Great Barrier Reef and the coast of Queensland, Australia. Numbers represent surveyed reefs, where; 1 = Stucco Reef, 2 = Channel Reef, 3 = Arlington Reef, and 4 = reefs surrounding Lizard Island (North Reef, South-Palfrey Reef, Loomis Reef, Casuarina Reef).

A further 146 colonies were collected and measured (maximum diameter, width, weight, species name, growth form) from the wild to complement data from commercially harvested coral species, focussing on expanding the size range of whole coral colonies to improve application of size-weight relationships to coral transect data. Divers on SCUBA searched for unattached colonies that were temporarily removed from the water to be measured and weighed before returning to the approximate position from where they were collected. Excess substrate was typically removed prior to weighing for both industry supplied and additional collected coral colonies. Any colonies where the base could not be removed were placed into the ‘colony with base’ category of the type variable.

To estimate standing biomass of *Acropora* at select sites, 41 video transects were recorded across 3 regions of the GBR (‘Central’, ‘Northern’, ‘Far Northern’) encompassing 7 reefs and 12 sites (a minimum of 100 m apart) recorded between August 2018 and November 2020 (Figure 3.1). For each transect, a 50-m transect tape was deployed along a depth contour between 1 and 12 m deep. A GoPro Hero 7 or 8 mounted on a camera jig was used for recording. The jig was assembled using 1.2. x 1.0 m PVC conduit poles in a ‘T’ configuration, with the camera mounted on the 1.2-m pipe at a 90° angle to the 1-m piece. The end of the 1-m pipe was placed on, or as close as practicable to the transect line and/or coral while swimming from one end of the transect line to the other (giving recorded belt transects a dimension of 50 x 1 m). Care was taken to maintain a constant camera angle (~65° angle to the plane in which diameter was measured), as well as avoid damage to corals and other benthic biota during surveys.

3.2.2. Processing and analyses

Size-weight relationships have been well described by power functions in other corals (Pratchett et al., 2020a), where coral weight (in the current case defined as ‘*W*’, g) is equal to a constant scaling factor (‘a’) multiplied by the corresponding coral diameter (‘*D*’, cm), to the power of a constant exponent (‘b’). A Bayesian nonlinear regression approach was used to model the size-weight relationship of *Acropora* colonies and fragments in R (R Core Team, 2020), following a lognormal (ln) distribution, with the response term ‘W’ representing weight in grams (see Equation 3.1). This was accomplished using the ‘brm’ function within the R package ‘brms’ (Bürkner, 2017a). In the current application, the non-linear predictor (‘η’) for the primary parameter ‘μ’ can be described as Equation 3.2, with the covariate (*D*, i.e. maximum diameter in cm) and the nonlinear parameters ‘a’ and ‘b’, each of which represent an intercept only predictor parameter and are therefore calculated as constants. The term *f* defines the structure of the user supplied function (equation 3), which is in this case a two-factor power function. For further explanation of distributional non-linear modelling in brms see Bürkner (2017a).

$$\boldsymbol{W_i \sim \ln(\mu_i, \sigma_i).} \qquad \textbf{Equation 3.1}$$

$$\boldsymbol{\eta_\mu = f(D, a, b)} \qquad \textbf{Equation 3.2}$$

$$\boldsymbol{f = aD^b} \qquad \textbf{Equation 3.3}$$

Bayesian statistical inference was performed via the Hamiltonian Monte Carlo algorithm and its extension No-U-Turn Sampler (NUTS) using the modelling software ‘Stan’ to model non-linear relationships, estimate the parameters of the user supplied power function, and investigate biomass estimation results using the R package brms. Efficient approximate leave-one-out (LOO) cross validation for Bayesian models via Pareto smoothed importance sampling (PSIS; Vehtari et al., 2017) was conducted to establish the validity of the non-linear power size-weight model and its predictive capacity using the function ‘loo’ within the package ‘brms’. This was accomplished via visual inspection of model and model parameters, and LOO comparison of potential distributions (lognormal and normal) and relationships (linear, power, exponential) between response and predictor variables. For a description of validation results and other details related to each model, see Appendix B, Table B1. Results are reported in terms of the elapsed difference (elpd_diff) and standard error difference (se_diff) in relation to the next best performing model.

Previous work describing size weight relationships in corals (Pratchett et al., 2020a) was used to inform the initial values of set priors, although further modification of these priors was applied congruent with model validation. To determine the suitability of combining visually similar growth forms (i.e., corymbose and caespitose, as well as staghorn and bottlebrush forms) into growth form groups, the effect of growth form on size-weight relationship was assessed using a non-linear model where the intercept and slope of each growth form group was allowed to vary (i.e., often termed a ‘random slope’ model) following the already described non-linear power relationship. Similarly, the potential influence of the type category (fragment, colony, colony with base) on *Acropora* size-weight relationship was also of interest, and so was included as a random slope category in analysis. Hypotheses regarding the statistical similarity of these growth forms were tested using the ‘hypothesis’ function in the ‘brms’ package. The non-linear power relationship model (containing all size-weight data) was compared to a series of random slope non-linear power relationship models containing the interaction of random slope category ‘growth form group’ and random slope category ‘type’, and a model each containing only one of the categories.

To obtain size (i.e., diameter) estimates of *Acropora* colonies from the transect surveys, video frames were captured and analysed using the ‘set scale’ and ‘measure’ tools in the software ImageJ (Schneider et al., 2012). The camera jig was used for scale. Average colony diameter was estimated for each coral. All *Acropora* colonies were then categorised into one of 6 growth forms (bottlebrush, caespitose, corymbose, digitate, staghorn and table) and later combined into new growth form groups (corymbose/caespitose, digitate, staghorn/bottlebrush, table). The biomass in grams of each coral was then calculated using the model equation derived from the estimated parameters of the user supplied function (Equation 3.2). Parameters were retrieved using the 'as_draws_rvars' function of the package 'posterior' (Bürkner et al., 2021).

To transform biomass estimate data into a biomass per unit area estimate, biomass for each growth form group was averaged to the transect level and divided by total transect area (50 m^2). A Bayesian ANOVA and pairwise comparison using the brms and 'emmeans' (Lenth, 2021) packages under a lognormal distribution was then performed on a dataset containing all biomass per m^2 data including the unvarying factors 'region' ('Central', 'Northern', 'Far northern') and 'growth form group' ('corymbose/caespitose', 'digitate' 'staghorn/bottlebrush', 'table') and the varying factors 'site' (12 sites in total) nested within 'reef' (see Figure 3.1 for list of reefs). For a full list of model formulas, priors, iterations, and other details related to each model, see Appendix B, Table B1. All figures and analyses were produced using the statistical software R v4.0.3 (R Core Team, 2020). R code available via Pacey (2024).

3.3. Results

3.3.1. Size-weight relationship models

A total of 3,972 coral pieces (including fragments and whole colonies) were used to establish size-weight relationships across four distinct growth form groups (corymbose/caespitose, digitate, staghorn/bottlebrush, table). For the data supplied to the size-weight models (i.e., including supplementary size-weight data), the average piece of *Acropora* had a diameter of 12.2 cm (± 0.11 *SE*) and weighed 301.0 g (± 16.70 *SE*). Bottlebrush *Acropora* made up 150 of the 508 data points for the staghorn/bottlebrush model, while caespitose growth forms made up 1048 of the 2619 datapoints for the corymbose/caespitose model (Figure 3.2). Average diameters did not appear to differ greatly between growth forms. Collected table *Acropora* pieces had the largest maximum diameter on average, reaching 15.20 cm (± 0.42 *SE*), followed by digitate (14.41 cm ± 0.50 *SE*), staghorn/bottlebrush (12.10 cm ± 0.40 *SE*); and corymbose/caespitose (11.30 ± 0.40 *SE*). Digitate growth form pieces tended to be the heaviest with an average weight of 883.0 g (± 175.00 *SE*), then table *Acropora* at 544.0 g (± 61.10 *SE*), while staghorn/bottlebrush and corymbose/caespitose had similarly low weights at 250.0 g (± 36.10 *SE*) and 200.0 g (± 7.61 *SE*), respectively. Data obtained from fishery sources only (i.e., excluding supplementary sampled data) followed the same group order as size-weight model data, although the average weight per growth form was noticeably lower and diameters were slightly lower (Table 3.1). It should be noted that the data described in Table 3.1 is not intended to be a representation of the proportional harvest of coral pieces and coral growth forms from the QCF, as it spans only two operators across two regions from a limited time period.

Table 3.1. *Acropora* pieces collected by the fishery for each growth form group (corymbose/caespitose, digitate, staghorn/bottlebrush, table), showing the number (n), mean ± *SE*, minimum value, and maximum value for (a) weight and (b) diameter.

Growth Form	**n**	**mean ± *SE***	**min**	**max**
a) Weight (g)				
Digitate	214	232.09 ± 16.38	42	2,666
Table	545	206.26 ± 10.70	11	4,485
Corymbose/Caespitose	2,593	173.02 ± 3.24	4	3,900
Staghorn/Bottlebrush	474	80.40 ± 3.50	8	878
b) Diameter (cm)				
Table	545	12.64 ± 0.15	3	48
Digitate	214	11.76 ± 0.23	3	28
Corymbose/Caespitose	2,593	11.10 ± 0.06	2	45
Staghorn/Bottlebrush	474	10.02 ± 0.14	3	23

Non-linear hypothesis testing provided evidence to suggest that the interaction between diameter (cm) and weight (g) was statistically equal for staghorn and bottlebrush growth forms (hypothesis test for class b, estimate±*SE* = 0.00 ± 0.02, evidence ratio = 93.71, posterior probability = 0.99), and so these groups were combined. There was only weak evidence to suggest the similarity of the corymbose and caespitose growth form groups (hypothesis test for class b, estimate±*SE* = 0.19 ± 0.08, evidence ratio = 0.52, posterior probability = 0.34). Upon visual inspection, the plotted model estimate for the caespitose group ran above the corymbose/caespitose group, although both model lines occurred within the others 95% probability band. However, as it can be difficult to accurately differentiate corymbose colonies from caespitose colonies via video transect, these groups were also combined in an effort to increase methodological robustness.

Both the type and growth form group category had an effect on size-weight relationships. The non-linear size-weight relationship model with both random slope ‘growth form’ and ‘type’ categories outperformed the models with either growth form or type only, as well as the simplistic model. As one of the core goals of this study is to provide tools to aid in bridging quantification between fishery data and environment, the decision was made to continue developing growth form group specific models emphasising simplicity, useability, and robustness of the final equation to data inputs via encompassing both coral fragments and colonies; although we acknowledge the type category (i.e., whether the coral piece is a fragment, a colony, or a colony with a base) also appears to impact the size-weight relationship of coral growth forms to some extent.

The size-weight relationships for each growth form appeared quite different, with some, such as digitate, showing a rapid increase in weight over a relatively short size range; and others, such as the staghorn/bottlebrush model, displaying a comparatively slow increase in weight over a larger diameter range (Figure 3.2). In all instances, the models are heavily driven by the smaller end of the maximum diameter scale, with the majority of pieces falling below 20 cm for all growth forms. The corymbose/caespitose and digitate models appear to be relatively good fits ($R^2 = 0.660$ and 0.729, respectively), with some overlap between the model 95% probability band and the sparse points towards the right side of the maximum diameter axis (Figure 3.2). The staghorn/bottlebrush ($R^2 = 0.718$) and table ($R^2 = 0.742$) models appear to overestimate weight for larger corals (> 45 cm and 60 cm, respectively), however this could be a sampling artefact related to the difficulty of collecting and weighing colonies of greater dimension *in situ*.

The estimated equation (broken grey line) closely follows the estimated model in all instances, occurring within the 95% probability band of each model. This equation is generated from the model estimates of the non-linear constant parameters (*a* and *b*) supplied to brms through the user-defined modelling function. While the generated models (coloured model lines) should be considered more robust, the provided equations represent effective approximations of these models, and do not limit their use only to those familiar with brms and R.

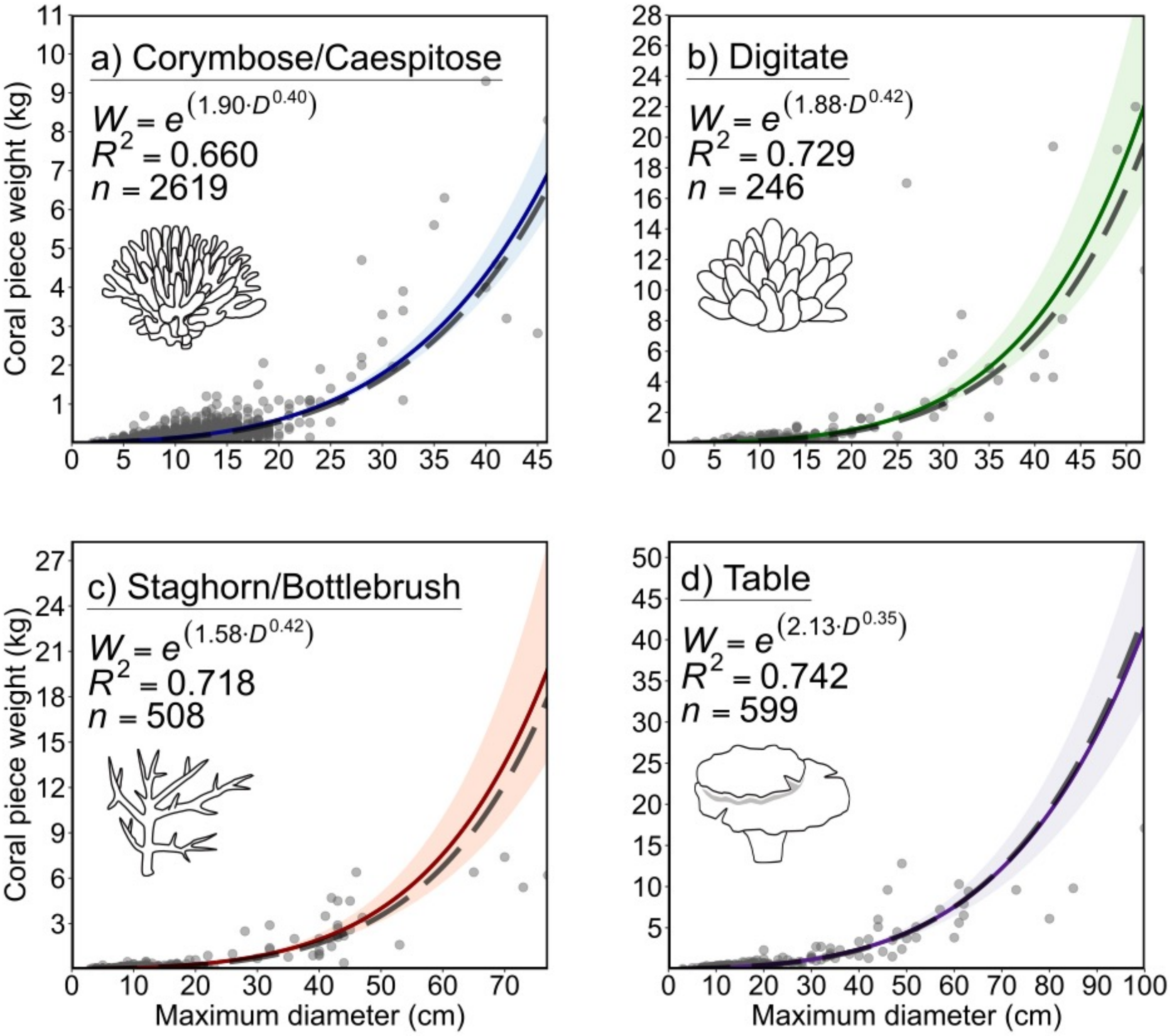

Figure 3.2. Modelled relationship between maximum diameter (cm) and weight (kg) of four Acropora growth form groups: (a) corymbose/caespitose, (b) digitate, (c) staghorn/bottlebrush, and (d) table. Probability band of the model (0.95) is indicated by the coloured bands surrounding model line, while grey dots indicate individual datapoints, and dashed grey model line represents the displayed approximated power relationship equation. For a full list of non-linear parameter constant estimates, SE, and High Density Intervals see Appendix B, Table B1.

3.3.2. Estimated biomass and region-specific biomass

A total of 9,220 *Acropora* colonies were measured from 41 (50 m x 1 m) video transects across three regions, including 3,189 (~35%) corals in the 'Central' region, 2,039 (~22%) corals in the 'Northern Central' region, and 3,992 (~43%) colonies in the 'Far Northern' region. Trends in size-weight data provided by fishery operators did not match the growth form prevalence patterns recorded from transects in the field (c.f. Table 3.1 and 3.2). Digitate corals occurred most frequently at sampled sites (n = 4571; ~50%), accounting for just under half of all recorded colonies, followed by table (n = 3221; ~35%), corymbose/caespitose (n = 1051; ~11%), and staghorn/bottlebrush (n = 377; ~4%).

Based on size-weight conversions, the total estimated standing biomass of *Acropora* corals across all 41 50 x 1 m video transects (pooling corals from all four growth categories and regions) was 65.22 $kg \cdot m^{-2}$, with transects ranging from 0.12 $kg \cdot m^{-2}$ to 4.70 $kg \cdot m^{-2}$. Staghorn/bottlebrush and corymbose/caespitose growth forms had the highest average estimated biomass and average diameter (Table 3.2). Digitate recorded the third highest average biomass and lowest average diameter, while table had the lowest average biomass and third highest diameter (Table 3.2).

Table 3.2. Estimates of sample size (n), mean ± SE, minimum value, and maximum value for (a) biomass and (b) diameter for each growth form (corymbose/caespitose, digitate, staghorn/bottlebrush, table) based on field transect data.

Growth Form	**n**	**mean ± *SE***	**min**	**max**
a) Biomass estimate (g)				
Staghorn/Bottlebrush	377	925.77 ± 103.96	8.97	18,968.88
Corymbose/Caespitose	1051	473.59 ± 30.29	12.51	13,403.65
Digitate	4571	282.93 ± 8.16	10.74	14,111.12
Table	3221	253.62 ± 7.88	13.93	9,838.65
b) Diameter (cm)				
Staghorn/Bottlebrush	377	23.23 ± 0.69	2.12	75.75
Corymbose/Caespitose	1051	14.80 ± 0.25	2.04	55.95
Table	4571	12.92 ± 0.14	1.91	67.96
Digitate	3221	10.68 ± 0.08	1.70	46.79

Analysis suggests that ‘reef’, ‘site’, ‘region’, and ‘growth form’ and the interactions between these factors all contribute to the variation observed in estimated *Acropora* biomass. The out-of-sample predictive accuracy of the ANOVA model containing all varying (reef and site) and non-varying (region, growth form) factors was significantly higher than all alternative models (nearest elpd_diff = -7.7, se_diff = 7.7, region only model) including a null model providing inferential evidence for the effect of these factors and their interactions. Average biomass ($g{\cdot}m^{-2}$) was highest in the northern region sample, followed by the far northern sample, and finally the central region sample (Figure 3.3), with only weak evidence (ANOVA with pairwise comparison, all combinations P<0.95) to suggest any region differed statistically in terms of average biomass ($g \cdot m^{-2}$). There was strong evidence (ANOVA with pairwise comparison, $P>0.95$) to suggest table *Acropora* have less biomass ($g \cdot m^{-2}$) compared to the corymbose/caespitose group across regions (Figure 3.3). Within regions, with the exception of the central region, a pattern emerged whereby the staghorn/bottlebrush group typically had greater biomass ($g \cdot m^{-2}$) than other groups with the exception of the corymbose/caespitose group, while digitate and table were more similar in biomass but typically not statistically lower than the corymbose/caespitose group (Figure 3.3).

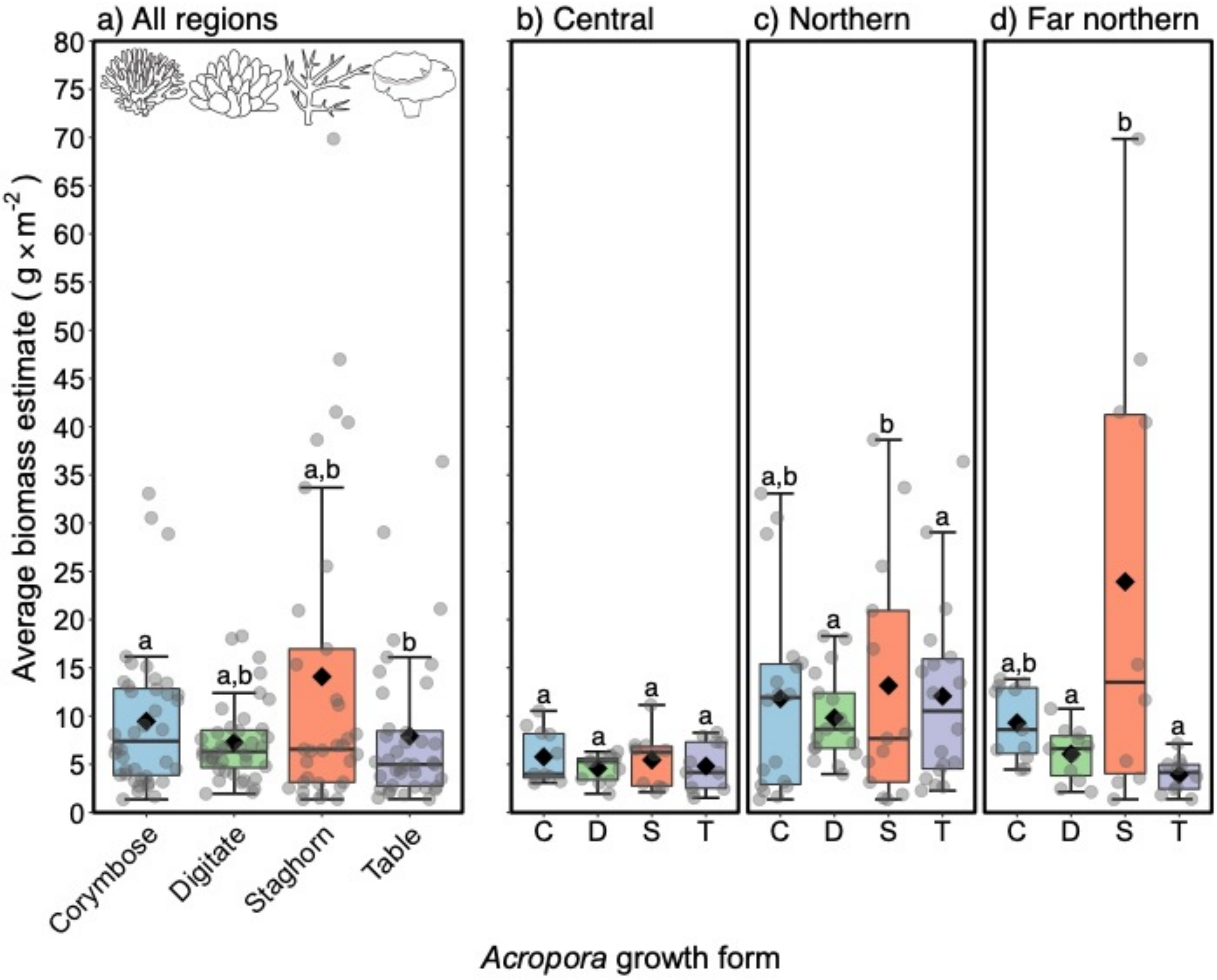

Figure 3.3. Average estimated biomass (g·m^{-2}) of coral for each growth form group: Corymbose (C) = corymbose/caespitose, Digitate (D), Staghorn (S) = staghorn/bottlebrush, and Table (T) from across (a) all regions, (b) Central region, (c) Northern region, and (d) Far northern region. Solid line represents median biomass (g·m^{-2}) estimate for growth form group, black diamond equals mean biomass (g·m^{-2}) estimate for growth form group, lower and upper hinges represent 25th and 75th percentiles (respectively), and whiskers estimate 95% confidence interval (*CI*). Different letters above whiskers represent significant differences between groups within each region plot.

3.4. Discussion

Establishing size-weight relationships for targeted *Acropora* growth forms is an important step towards assessing the potential ecological and environmental impacts of current harvest levels and practices in coral harvest fisheries globally. Size-weight relationships for corymbose/caespitose, digitate, staghorn/bottlebrush, and table growth form groups were effectively described using a scaling power relationship with a constant exponent, providing a means for estimating weight from a quickly and easily obtained measurement (i.e., diameter). Through the application of the described methodology in the study region, size-weight relationships were mathematically approximated, allowing for calculation of biomass from transects conducted in the QCF. An important implication of this relationship is that larger colonies contribute disproportionately to the standing biomass of all coral types. Considering biomass in terms of weight per colony, a single coral in the maximum size range of biomass transect data (75-80 cm) is equivalent to 599 corals in the smallest <5 cm size range, and around 97 corals in the 10-15 cm range. This was most extreme in the staghorn/bottlebrush group (2115 corals for <5 cm category, 187 for 10-15 cm; maximum size range 75-80 cm), followed by corymbose/caespitose (467 corals for <5 cm, 74 for 10-15 cm; maximum size range 55-60 cm), digitate (419 corals for <5 cm, 56 for 10-15 cm; maximum size range 45-50 cm), with table (303 for <5 cm, 63 for 10-15 cm; maximum size range 65-70 cm) showing the least extreme relationship. In light of this, resilience to harvesting for *Acropora* spp. will be greatly enhanced by taking relatively small fragments from established colonies rather than removing entire *Acropora* colonies. Aside from the disproportionate contribution to biomass from larger colonies, this is most likely true even in consideration of potential costs to donor colonies (Lirman, 2000; Wallace, 1985). Therefore, practices favouring the collection of small fragments (thus taking greatest advantage of the modular nature of *Acropora* corals) should be encouraged.

3.4.1. Composition of sample coral catch

Acropora coral specimens provided by Queensland coral collectors were predominantly fragments (82%), typically around ~12 cm in diameter, and were mostly composed of corymbose/caespitose growth forms (~68%, Table 3.1). Staghorn/bottlebrush corals had the highest proportion of fragments compared to entire colonies at ~98% fragments, followed by table (~85%), digitate (~83%) and corymbose/caespitose (~78%). In contrast to coral collections, the predominant (~50%) *Acropora* corals recorded on transects were small (~11 cm) digitate colonies (Table 3.2). While coral collections were not necessarily representative of overall catch (across multiple collections by different licence holders) and the localities of these collections were not exactly matched to the sites where video transects were run, apparent differences in composition between coral collections and field-based assessments underline the highly selective nature of coral harvesting collections for the marine aquarium industry (see also Harriott, 2003; Wood et al., 2012). On one hand, this selectivity (which is based not only on species identity, but also colony colour, size and shape) protects against ecological depletion of corals. On the other hand, highly selective harvesting may also drive shifts in coral composition (Bruckner and Borneman, 2006), with potential for localised depletion of heavily harvested species, especially if they are generally rare. The ecological impact of such selective harvesting would require a much more detailed assessment of harvest practices, relating local levels of extraction to specific availability of individual species. The size-weight conversion models generated here will help bridge this gap and enable comparisons between species-specific harvest levels and standing biomass.

3.4.2. Standing biomass calculations and implications

Estimates of standing biomass presented herein, together with habitat mapping throughout the entire Great Barrier Reef Marine Park (Roelfsema et al., 2018) allow for provisional estimates of standing biomass, albeit for groups of species (cf. individual species). As an example, we focus on one location in the current study, Stucco Reef in the

Central GBR. The average estimated biomass of *Acropora* spp. (all species and growth forms) at Stucco Reef, based on 12 video transects across 4 distinct sites (with total sample area of 600 m^2) was 22.46 $g \cdot m^{-2}$, with a mean of 5.85 $g \cdot m^{-2}$ for corymbose/caespitose *Acropora*. If we extrapolate this to the entire shallow reef area of Stucco Reef (~4.74 km^2) the estimated harvestable biomass of *Acropora* would be over 106 t, with over 27 t of corymbose/caespitose *Acropora*. This represents ~76% (just under ~20% for corymbose/caespitose alone) of the 140 t TACC for "other corals", which could theoretically comprise entirely *Acropora* species. In actuality, coral harvesting is spread across multiple reefs, to the extent allowed by management regulations and operational constraints. Although current harvest levels and limits at the scale of the entire GBR likely represent much less than 1% of the entire standing biomass of *Acropora* corals (see Harriott, 2001), there are additional complexities to consider in the maintenance of a sustainable harvest strategy.

While the considerable standing biomass of *Acropora* spp. corals available to collectors operating on the QCF may buffer the environmental impacts of local scale coral harvesting, the vulnerability (Baird and Marshall, 2002; Hughes et al., 2018b; Muko et al., 2013) of these structurally important corals to large-scale disturbances (including population irruptions of crown-of-thorns starfish, severe cyclonic storms and climate-induced bleaching) does have the potential to undermine the sustainability of commercial coral collection. *Acropora* corals are likely to be comparatively resilient to collection, being among the fastest growing coral species (Pratchett et al., 2015), especially compared to some of the other major species targeted by global aquarium fisheries (Bruckner and Borneman, 2006; Petrossian et al., 2020; Wood et al., 2012). Additionally, the removal of small fragments (instead of entire colonies) is likely to further limit the direct ecological impacts of collection. However, *Acropora* is particularly vulnerable to acute regional disturbances such as cyclones and mass coral bleaching events (Baird and Marshall, 2002; Hughes et al., 2018b; Muko et al., 2013) and the frequency and/ or intensity of such disturbances is set to increase in coming years (Cheal et al., 2017; Hughes et al., 2018a). *Acropora* spp. are also a major structural component of the reef landscape (Coker et al., 2014), providing habitat for coral reef fishes and the myriad species that dwell on or make use of coral reefs, thus also providing valuable ecosystem services (Pratchett et al., 2014).

Demonstrating that coral fisheries are ecologically, economically, and socially sustainable is very important on a number of fronts. First and foremost, establishing and implementing sustainable harvest limits for all individual species of CITES-listed corals will assure positive NDF (i.e., non-detriment finding) determinations, which are essential for continued global trade in hard corals (Dee et al., 2014). Secondly, it is also vital that the independent threat posed by coral harvesting, especially given escalating disturbances and human pressures that are causing sustained and widespread degradation of coral reef ecosystems is well understood in a management context. Given the plight of coral reefs globally, aquarium retailers and buyers are favouring corals (and other traded marine biota) of certain types and/ or sourced from specific locations, where there are little or no sustainability concerns (e.g., Rhyne et al., 2012; Fraser et al., 2021).

3.4.3. Conclusions

Establishing fisheries sustainability globally is fundamentally dependent on rigorous and ongoing stock assessments, to relate harvest levels and limits to changes in abundance (and more critically for corals, biomass) of individual species or species groups. This study established size-weight relationships for the most commonly occurring *Acropora* growth forms to allow quantification of biomass per unit area within three major regions of the QCF. This is an important step towards reconciling ecological and environmental impacts of coral fisheries operating across the GBR and across the world. Although the direct and independent risk posed by harvesting across all *Acropora* spp. may be comparatively small, harvest levels and limits need to be carefully considered given the ecological importance and vulnerability of these corals, which also varies among the different species.

Chapter 4

Using size-weight relationships to estimate biomass of heavily targeted aquarium corals by Australia's coral harvest fisheries[2]

4.1. Introduction

Marine aquarium fisheries supply an international trade involving the collection and sale of millions of live organisms (Wood et al., 2012), many of which are collected from the wild in countries such as Indonesia, Fiji, and Australia (Rhyne et al., 2014; Wood et al., 2012). Coral harvest fisheries supply a major component of the aquarium trade in the form of live aquarium coral specimens, the vast majority of which are destined for home or public aquaria (Dee et al., 2014; Rhyne et al., 2014). Coral harvesting for the marine aquarium trade is highly selective, and involves hand collection of select coral colonies or fragments (Harriott, 2001; Rhyne et al., 2012). Therefore, the direct and independent ecological impacts of coral harvest fisheries are generally regarded as very minor (e.g., Harriott, 2001), especially compared to the myriad of large-scale disturbances (e.g., outbreaks of coral predators and climate-induced bleaching) that are affecting wild coral stocks. However, the widespread and accelerating degradation of coral reef ecosystems (Bruno and Valdivia, 2016; Hughes et al., 2017b) and coastal modification (Fabricius, 2005; Wear, 2016) may undermine the sustainability of coral harvest fisheries. In light of the already concerning state of coral reef

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environments globally, industries that access or use coral reef environments, such coral harvest fisheries, and the governmental and intergovernmental bodies that manage them, are currently experiencing significant and increasing public, economic, and political pressure to deliver scientifically defensible and ecologically relevant management policies (Burke et al., 2011; Hughes, 1994; Negri et al., 2002).

All hard corals are listed in Appendix II of the Convention on the International Trade in Endangered Species (CITES), and strict stipulations must be met for the trade of listed wildlife to be considered legal in all 184 signatory countries. To supply the growing international demand for live aquarium coral species, hundreds of thousands of coral pieces are traded annually valued at millions of dollars. Wood et al. (2012) examined global import reports from the CITES trade database, noting marked increases in the volume of trade from ~600,000 pieces in 2000 to ~990,000 pieces in 2010, peaking at >1,500,000 pieces in 2007 (see also Rhyne et al., 2012, Dee et al., 2014). International trade in live corals further increased in 2016-2018 (Food and Agriculture Organization of the United Nations, 2021). Prior to 2018, Indonesia and Fiji were the largest exporters of hard (order Scleractinia) corals (i.e., accounting for 70.0% and 10.3% of global exports, respectively; Wood et al., 2012), but both these countries imposed significant constraints on wild coral harvesting (in 2017-2018) due to concerns of overharvesting, which significantly changed the nature of global coral trade.

Environmental concerns regarding coral harvest fisheries have mostly focused on the potential for localized depletion of highly vulnerable or heavily targeted coral species (Bruckner and Borneman, 2006; Harriott, 2001). Many of the most heavily targeted coral species have large fleshy polyps (often referred to as Large Polyp Stony corals; LPS corals), which are presumed to be slow growing and long lived (Bruckner and Borneman, 2006).

Despite a lack of relevant biological information for many of these species, it is presumed that such species will be highly vulnerable to over-fishing (Bruckner and Borneman, 2006; Rhyne et al., 2014). Particular concern also exists where fisheries exploitation is compounded by rapid and accelerating environmental change (Bruckner and Borneman, 2006).

In Australia, commercial coral harvest fisheries are managed at the state/territory level with major fisheries in Queensland (Queensland Coral Fishery; QCF), Western Australia (Western Australian Marine Aquarium Fish Managed Fishery; WAMAFMF), and the Northern Territory (Northern Territory Aquarium Fishery; NTAF). The QCF operates over a total area of 24,000 km^2 in permitted zones of the Great Barrier Reef Marine Park (GBRMP). The QCF has the largest annual Total Allowable Commercial Catch (TACC) of all Australian coral fisheries at 200 t, which is split between 60 t of ‘specialty’ or LPS corals and 140 t of ‘other’ corals; which includes branching taxa such as the Acroporidae (Pratchett et al., 2020a). Concerns have previously been raised regarding the potential for the localised depletion of specific coral species in areas of concentrated fishing activity (Jones, 2011), combined with the threat posed by extrinsic disturbances such as cyclones, outbreaks of crown-of-thorns starfish and mass coral bleaching (De’ath et al., 2012; Pratchett et al., 2020a, 2020b). The WAMAFMF is the second largest coral fishery. It is a low volume and high value fishery that operates over a total gazetted area of 20,781 km^2 along Australia’s west coast, with a TACC of 15 t for hard and soft corals. The NTAF operates in all inland, estuarine, and marine waters to the outer boundary of the Australian Fishing Zone (AFZ) in NT waters, incorporating an area of 523,946 km^2 of marine habitat. This fishery has the only taxon-specific quota levels, typically assigning 80 kg for individual species, and 160 kg for species groups.

A recent re-assessment of the QCF (Pratchett, 2021a), as part of the necessary process of seeking Wildlife Trade Operation (WTO) approval from CITES, has highlighted the growth in this fishery and the continued prominence of LPS corals since the last comprehensive assessments of global import and export trends in the trade of ornamental corals (Rhyne et al., 2012; Wood et al., 2012). This re-assessment examined QCF catch trends from the 2006/2007 to 2019/2020 financial years, with a similar re-assessment now underway for the WAMAFMF and NTAF as WTO approval expires late in 2022. Importantly, while *Acropora* corals form a major and increasingly large proportion of the QCF coral harvest (Pratchett, 2021a), LPS species remain an important component both in volume and economic contribution (Wood et al., 2012). The three major LPS coral families (Lobophylliidae, Merulinidae, Euphylliidae) together accounted for ~41% of the QCF's total catch composition in the 2019-2020 financial year. Furthermore, there are 6 species that are of particular economic, biological, and ecological interest, namely *Homophyllia* cf. *australis* (Milne Edwards & Haime, 1848; Lobophyllia), *Micromussa lordhowensis* (Veron & Pichon, 1982; Lobophylliidae), *Catalaphyllia jardinei* (Saville-Kent, 1893; Merulinidae), *Trachyphyllia geoffroyi* (Audouin, 1826; Merulinidae), *Euphyllia glabrescens* (Chamisso & Eysenhardt, 1821; Euphyllidae), and *Duncanopsammia axifuga* (Milne Edwards & Haime, 1848; Dendrophylliidae). Together, these species accounted for over 35% of 2019-2020 total catch composition for the QCF, representing a total of 14.2 t or 317,718 coral pieces (Pratchett, 2021a). The Ecological Risk Assessment category for some of these heavily targeted LPS corals has recently been upgraded in the Queensland Coral Fishery, with *H.* cf. *australis* and *M. lordhowensis* now considered to be at "extreme" risk of experiencing an "undesirable event" as a result of fishery actions, while *E. glabrescens* and *T. geoffroyi* were upgraded to the "high risk" category (Morton et al., 2022).

There are uncertainties regarding the sustainability of harvest levels and harvest limits for Australian coral harvest fisheries (Pratchett, 2021a), mainly because the status and trends for targeted coral species and stocks are largely unknown. Harvest logs for these fisheries are currently reported in terms of weight, however, it can be difficult to understand the ecological relevance of recorded catch from this metric alone. Fundamentally, most existing accessible data on the abundance of harvested coral species based on either coral cover or colony densities (e.g., those outlined in Mellin et al., 2021) are not useful for establishing the ecological context of weight-based harvest limits. Additionally, some corals are collected as fragments as opposed to entire individuals, causing further difficulties if attempting to utilise coral cover or abundance data for fisheries monitoring. Instead, estimates of harvestable or 'standing' biomass would provide a better unit of quantification for assessment of coral harvest quotas and the ecological impact of harvesting (Ross, 1984). To bridge this gap between harvest and ecological impact, Longenecker et al. (2015) utilised size-weight relationships to estimate standing biomass of *Acropora* corals, suggesting that establishing these relationships in other corals is likely to be a viable approach that can provide managers with a relatively simple methodology able to place fisheries harvests into an ecological context (see also Chapter 3; Pacey et al., 2022).

To facilitate improved management of Australia's coral harvest fisheries (and perhaps internationally), this study modelled the relationship between maximum diameter and coral weight for six key LPS coral species: *Catalaphyllia jardinei*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Homophyllia* cf. *australis*, *Micromussa lordhowensis*, and *Trachyphyllia geoffroyi*. Aside from providing a mechanism to calculate standing biomass of these coral species, establishing size-weight relationships for these corals will provide an opportunity to assess previously unexamined biological characteristics of these species relating to these described relationships such as whether these corals exhibit isometric or allometric growth patterns. Moreover, size-weight information is combined with *in situ* video transects to demonstrate the utility of this method for estimating standing biomass of individual corals.

4.2. Methods

4.2.1. Data collection

The size-weight relationships of six coral species (*Catalaphyllia jardinei*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Homophyllia* cf. *australis*, *Micromussa lordhowensis*, and *Trachyphyllia geoffroyi;* see Figure 4.1) was investigated using coral pieces provided by commercial collectors between March 2016 and July 2020. Samples were collected from Queensland, Western Australia, and the Northern Territory. Collection is typically far more geographically focused in the latter two states, and so locations were supposed to represent the major areas of operation in each state. Species selected for this study were prioritised based on their importance to coral fisheries exports across Western Australia, Northern Territory, and Queensland; and the perceived risk to these species in terms of overfishing and/or fishery independent threats (as identified in Pratchett et al., 2020a).

Figure 4.1. LPS (Large Polyp Stony) corals targeted for commercial collection used in the current study, namely; A) *Catalaphyllia jardinei,* B) *Duncanopsammia axifuga* C) *Euphyllia*

glabrescens D) *Homophyllia* cf. *australis*, E) *Micromussa lordhowensis*, F) *Trachyphyllia geoffroyi*.

Data used to generate size-weight relationship models for each species was collected in collaboration with fishery operators. The maximum diameter and perpendicular diameter of each coral was recorded to the nearest millimetre using callipers, or a ruler for larger corals, while weight was recorded to the nearest gram using an electronic scale. Corals were left to drain freely for 2-5 minutes prior to processing. Samples were mostly intact whole coral colonies (25 fragments total, <0.01% of total sample size), with excess substrate removed prior to weighing. Care was taken to also remove excess water prior to weighing of corals. In addition to size and weight measurements, site, region, and state from which the sample was obtained was also recorded.

Table 4.1. Summation of known biological traits of commonly harvested LPS coral taxa in Australia. For 'Reproduction' (i.e., reproductive traits); H = Hermaphroditic; G = Gonochoristic; B = Brooding; S = Broadcast spawning. For 'Coralite' (i.e., corallite form); P = Polystomatous; M = Monostomatous. 'Substrate' (i.e., substrate attachment) and 'Growth Form' listed are representative of mature corals, as most planula larvae settle and attach to a hard substrate and tend to initially develop in a solitary form. Information summarised from Pratchett et al (2020a) and Pratchett (2021).

Species	**Substrate**	**Growth form**	**Reproduction**	**Corallite**
Catalaphyllia jardenei	Free	Flabello-meandroid	H, S	P
Duncanopsammia axifuga	Attached/Free	Tabular branching	G, S	P
Euphyllia glabrescens	Attached/Free	Phaceloid	H, B	P
Homophyllia australis	Free	Solitary	H, S	M
Micromussa lordhowensis	Attached	Massive	H, S	P
Trachyphyllia geoffroyi	Free	Flabello-meandroid	H, S	M

To demonstrate the utility of modelled size-weight relationships, we estimated the standing biomass of the six focal species (*Catalaphyllia jardinei*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Homophyllia* cf. *australis*, *Micromussa lordhowensis*, and *Trachyphyllia geoffroyi*) across a total of 204 video transects conducted in two states (QLD, WA) across six locations (Cairns, Dampier, Exmouth, Karratha, Mackay, Southern Great Barrier Reef), 15 reefs, and 34 sites. Video transect surveys were conducted on SCUBA using a 50-m transect tape deployed along a depth contour ranging from 0 m to 20.9 m. A GoPro Hero 7 mounted on a camera jig was used to record the substrate along each depth contour. The camera jig was made using 1.2. x 1.0 m PVC conduit pipes, with the camera mounted on the 1.2m pipe perpendicular to the 1m piece in a ‘T’ configuration. The end of the 1m pipe was placed on, or as close as practicable to the transect line and/or coral while swimming from one end of the transect line to the other (giving recorded belt transects a dimension of 50 x 1 m). Care was taken to maintain camera angle, as well as avoid damage to corals and other benthic biota during surveys.

4.2.2. Processing and analyses

A Bayesian non-linear regression approach was used to model the relationship between maximum coral colony diameter in cm (*D*) and weight in g (*W*) within a gaussian distribution (see Equation 4.1). In this approach, the non-linear predictor (‘η’) for the primary parameter ‘μ’ can be described following Equation 4.2, with the covariate (‘*D’*, i.e., maximum diameter in cm) and the nonlinear parameters ‘a’ and ‘b’, each of which represent an intercept-only predictor parameter and are therefore calculated as constants. The term ‘*f’* defines the structure of the user supplied function (equation 3), which in this case is a two-factor power function. Size-weight relationships have been well described by power functions in corals previously (Chapter 3; Longenecker et al., 2015; Pacey et al., 2022), where coral weight (in the current case defined as ‘*W*’, g) is equal to a constant scaling factor (‘a’) multiplied by the corresponding coral diameter (‘*D*’, cm), to the power of a constant exponent (‘b’). For further explanation of the distributional non-linear modelling approach utilised in this study, see Bürkner (2017b).

$$\boldsymbol{W_i \sim N(\mu_i, \sigma_i)} \qquad \textbf{Equation 4.1}$$

$$\boldsymbol{\eta_\mu = f(D, a, b)} \qquad \textbf{Equation 4.2}$$

$$\boldsymbol{f = aD^b} \qquad \textbf{Equation 4.3}$$

Bayesian statistical inference was performed via the Hamiltonian Monte Carlo algorithm and its extension No-U-Turn Sampler (NUTS) using the statistical software Stan to model non-linear relationships, estimate the parameters of the user supplied power function, and investigate biomass estimation results. This was accomplished using the 'brm' function within the R[35] (ver 4.1.3) package 'brms' (Bürkner, 2017a). R code available via Pacey (2024). Efficient approximate leave-one-out (LOO) cross validation for Bayesian models via Pareto smoothed importance sampling (PSIS; Vehtari et al., 2017) was conducted to establish the validity of the non-linear power size-weight model and its predictive capacity using the function 'loo' within the package 'brms'.

Alternative models (e.g., linear, non-linear exponential) were investigated visually using the 'geom_smooth' function (method loess) in the package ggplot2 (Wickham, 2016) with decisions validated using the 'loo' function. Previous work describing size-weight relationships in corals (Pratchett et al., 2020a) was used to inform the initial values of set priors, with further modification of these priors applied where necessary during model validation. A Bayesian ANOVA (utilising median values) and pairwise comparisons were used to assess statistical difference in weight (g) and biomass per unit area ($g{\cdot}m^2$) and maximum diameter (cm) between species. To further compare the size-weight relationship between species, a lognormally distributed ANCOVA-type model was used. A pairwise comparison approach was utilised to compare the differences at the '*c*' non-linear parameter, with further investigation of one-sided posterior probability conducted using the package 'emmeans'. For a full list of priors see Appendix B, Table B1.

To measure the size of each target coral recorded during video transect surveys, video frames were captured and analysed using the 'set scale' and 'measure' tools in the software ImageJ (Schneider et al., 2012). The camera jig was used to set the scale of captured frames for measurement when the 1-m conduit was over or directly next to the target coral colony. A maximum colony length and colony width was then recorded for each coral. The biomass in grams of each coral was then estimated via the relevant species-specific size-weight relationship model equation derived from the estimated parameters of the user supplied function (Equation 4.2). To convert transect biomass estimate data into a biomass per unit area estimate, biomass for each growth form group was averaged to the transect level and divided by total transect area (50 m x 1 m). R code available via Pacey (2024).

4.3. Results

4.3.1. Size and weight trends

A total of 2,548 corals were measured across the six study species: *Catalaphyllia jardinei* (n = 43), *Duncanopsammia axifuga* (n = 219), *Euphyllia glabrescens* (n = 265), *Homophyllia* cf. *australis* (n = 436), *Micromussa lordhowensis* (n = 685), *Trachyphyllia geoffroyi* (n = 900). The majority of coral pieces were collected in Queensland (1,986 corals; ~78%), with ~21% (536) corals collected in Western Australia and ~1% (26) corals sampled in the Northern Territory. Due to uncertainties in taxonomy and range for the corals *Homophyllia* cf. *australis* and *Micromussa lordhowensis*, only corals from QLD were used to model data (Table 4.1).

Table 4.2. Coral pieces used to construct size-weight models showing the state that the samples were obtained from, number (n), mean ± *SE*, minimum (min) and maximum (max) value of samples by a) coral weight (g), and b) coral diameter (cm) for each species and state.

Species	State	n	mean ± *SE*	min	max
a) Weight (g)					
Catalaphyllia jardinei	QLD	43	166.19 ± 33.97	17	985
Duncanopsammia axifuga	WA	172	258.28 ± 25.55	10	2,750
	QLD	47	126.06 ± 15.05	17	483
*Euphyllia glabrescens**	WA	191	202.02 ± 18.67	6	1,530
	QLD	48	130.87 ± 22.47	16	948
	NT	26	24.96 ± 2.48	5	53
Homophyllia cf. *australis*	QLD	436	72.88 ± 2.12	8	448
Micromussa lordhowensis	QLD	685	314.06 ± 11.26	10	1,952
Trachyphyllia geoffroyi	QLD	727	249.95 ± 7.85	8	1695
	WA	173	93.60 ± 5.76	8	1,695
b) Diameter (cm)					
Catalaphyllia jardinei	QLD	43	7.52 ± 0.46	3.90	13.40
Duncanopsammia axifuga	WA	172	11.20 ± 0.37	2.60	30.50
	QLD	47	10.27 ± 0.54	4.50	20.70
*Euphyllia glabrescens**	WA	191	8.32 ± 0.25	2.10	19.20
	QLD	48	7.67 ± 0.46	3.20	17.60
	NT	26	4.28 ± 0.22	2.00	6.30

Homophyllia cf. *australis*	QLD	436	5.76 ± 0.05	2.50	9.75
Micromussa lordhowensis	QLD	685	11.48 ± 0.16	3.44	25.60
Trachyphyllia geoffroyi	QLD	727	9.20 ± 0.08	2.80	19.29
	WA	173	6.96 ± 0.10	3.70	10.60

*Some evidence to suggest *E. glabrescens* may represent different species across states (based on tentacle morphology, tank compatibility, planulation time, and size).

Australian coral fisheries tend to harvest small pieces or colonies of LPS corals, as shown by the mean diameter and weight of corals provided for the current study. *M. lordhowensis* corals had the highest average weight and diameter, followed by *D. axifuga*, *T. geoffroyi*, *E. glabrescens*, *C. jardinei*, with *H.* cf. *australis* having the lowest average weight (Table 4.1). There was substantial statistical evidence to suggest that all coral species differed in weight (Bayesian ANOVA pairwise comparison, P<0.01), with the exception of *C. jardinei* and *E. glabrescens* (Bayesian ANOVA pairwise comparison, P=0.341). Similarly, there was strong evidence to suggest a difference in average diameter between all corals (Bayesian ANOVA pairwise comparison, P<0.01), again with the exception of *C. jardinei* and *E. glabrescens* (Bayesian ANOVA pairwise comparison, P=0.308). For corals which were sampled from multiple states (*D. axifuga*, *E glabrescens*, *T. geoffroyi*), there was significant evidence (Bayesian ANOVA pairwise comparison, P<0.01) to suggest that all samples differed between states in terms of weight. Coral weight samples from QLD were significantly lower than those from WA for *D. axifuga*. For *E. glabrescens*, weight (g) of samples from the NT were lowest, followed by QLD, with WA being on average higher, all states being significantly different to each other. For *T. geofroyi*, corals sampled from QLD had significantly higher weight than those from Western Australia. For maximum diameter (cm), samples from the NT were significantly lower than samples from both QLD and WA, but QLD and WA were also statistically different. For *T. geffroyi*, the maximum diameter of samples from QLD was higher than samples collected from Western Australia.

4.3.2. Size-weight relationships

The relationship between size and weight for all individual species was well represented by a normally distributed non-linear power formula, which outperformed the alternative linear and exponential functions (LOO, closest elpd_diff = -348.8, se_diff = 19.7 for the linear alternative). However, the relationship between size and weight was statistically different among coral species (Pairwise comparison on 'c' parameter for ANCOVA type model, 95% HDP interval, median point estimate) with the exception of *C. jardinei* and *E. glabrescens*, which were the only two species which did not significantly differ in the 'c' parameter species coefficient ($P > 1 = 0.50$; Figure 4.2). All species therefore exhibit fundamental differences in their size to weight relationship over the measured diameter ranges. The contrast between *Homophyllia* cf. *australis* and *Trachyphyllia geoffroyi* was the only other contrast that did not receive a 0 or 1 probability; however, their posterior probability value is still considered highly significant ($P > 1 = 0.99$). From a fisheries perspective, this means that all species should be assessed and considered separately in terms of their size-weight relationship to properly estimate biomass and place the fisheries impact of each species into an appropriate, relevant, ecological context. Following this finding, the decision was made to model all species separately using an intercept only *a* and *b* parameter approach.

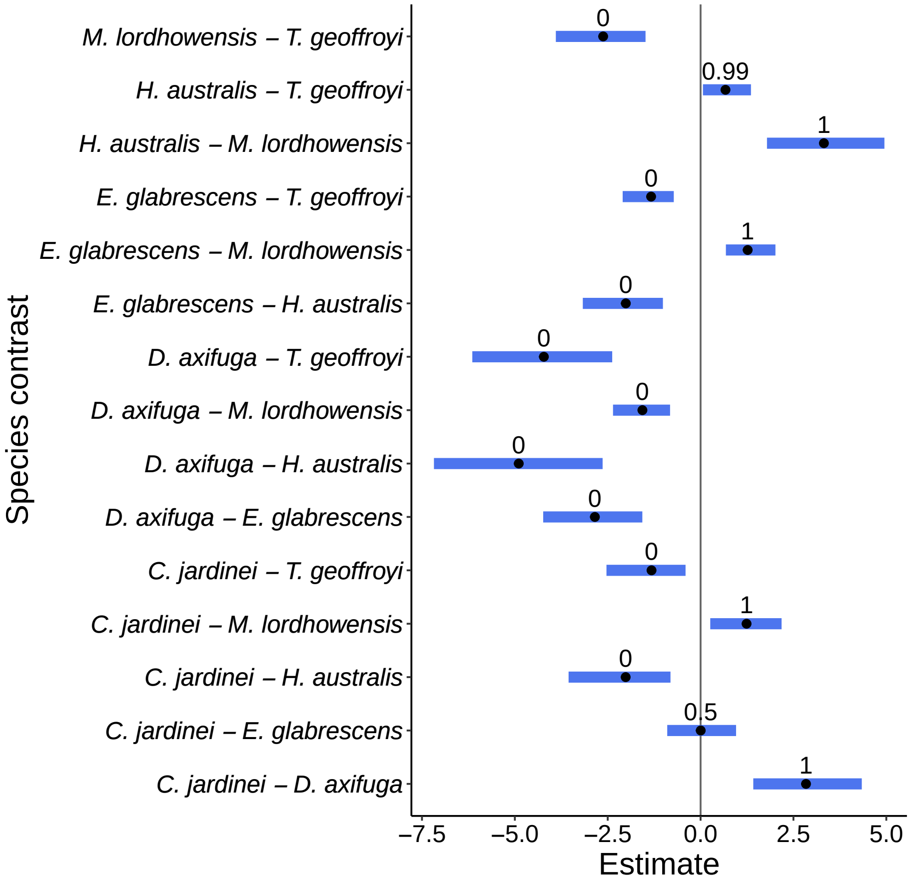

Figure 4.2. By species (*Catalaphyllia jardinei*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Homophyllia* cf. *australis*, *Micromussa lordhowensis*, *Trachyphyllia geoffroyi*) pairwise comparison of species coefficient estimate on the non-linear ‘c’ parameter, where blue bars represent 95% C.I. (credible interval), black point represents estimate of effect, and number above blue bar represents the probability that P > 1, where terms outside the 5-95% interval indicate statistical significance.

The relationship between coral diameter and weight as modelled in the current instance was able to account for the majority of variation in data across all species, with R^2 values ranging from 0.69 (*C. jardinei*) to 0.91 (*D. axifuga*) (Figure 4.3). The size-weight equation (broken grey line) was well estimated using the intercept only parameter approach, following the statistical model line (coloured line) and staying within the 95% confidence band in most instances, with the exception of *C. jardinei*. In the instance of the model line and the estimated equation, both appear to represent somewhat of an underestimate. This reflects greater error on the ‘a’ and ‘b’ non-linear intercept parameters, most likely resulting from sample size limitations.

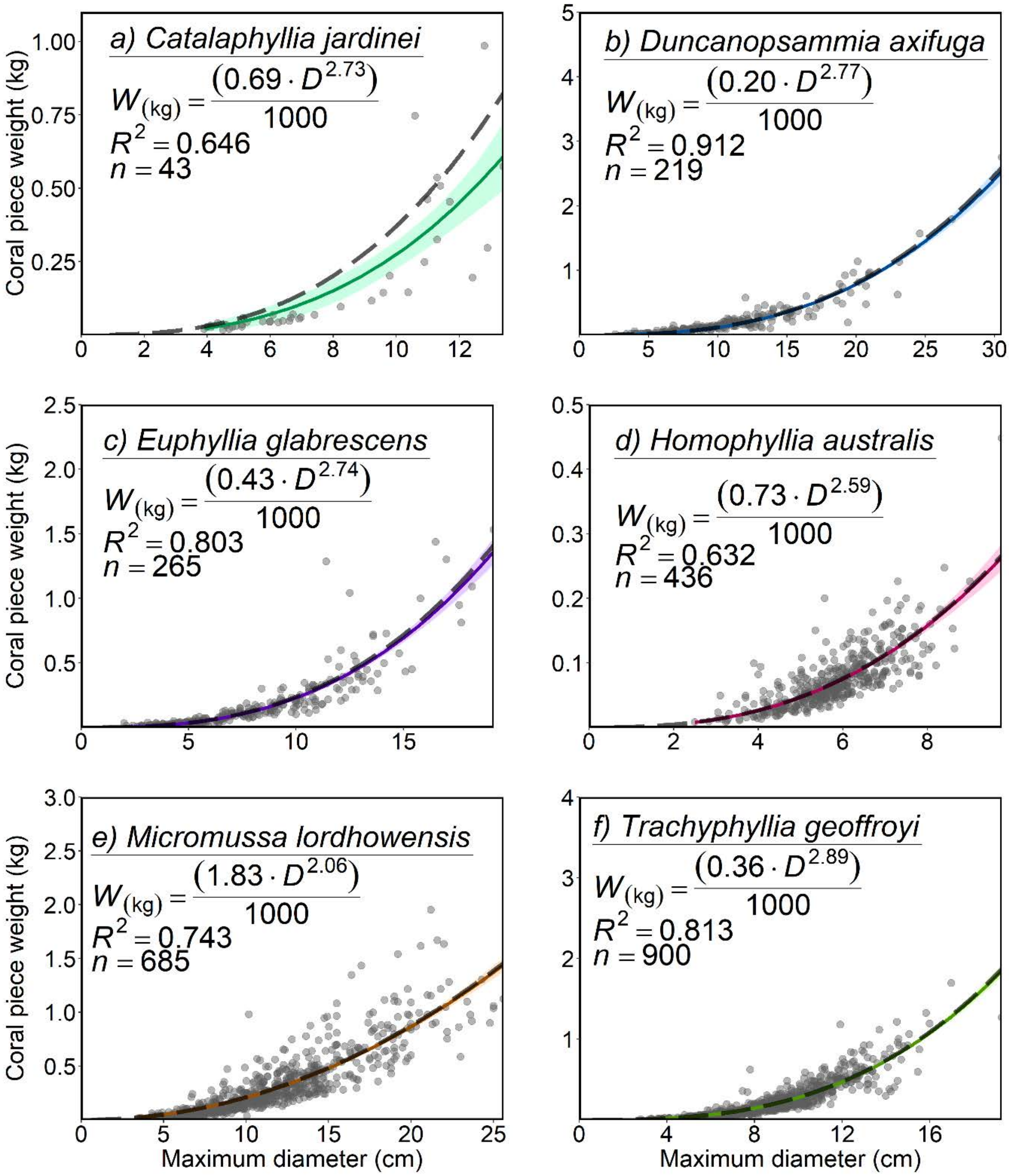

Figure 4.3. Modelled relationship between maximum diameter (cm) and coral weight (g) for the species: a) *Catalaphyllia jardinei*, b) *Duncanopsammia axifuga*, c) *Euphyllia glabrescens*, d) *Homophyllia* cf. *australis*, e) *Micromussa lordhowensis*, f) *Trachyphyllia geoffroyi*. Probability band of the model (0.95) is indicated by the coloured bands surrounding model line, while grey dots indicate individual datapoints, and dashed grey model line represents the displayed approximated power relationship equation. For a full list

of non-linear parameter constant estimates, SE, and High Density Intervals see Appendix C Table C1 & C2.

Model and equation lines typically bisect the major cluster of samples (e.g., *H.* cf. *australis*, *M. micromussa*), although there is evidence of underestimation in the higher maximum diameter range for some species (e.g., *M. micromussa*, *T. geoffroyi*). By all indications in the current study, all selected coral species with the exception of *C. jardinei* (2.73 ± 0.35 S.E., 95% C.I. 2.14 – 3.18, $P < 3 = 0.22$) appear to exhibit allometric growth, with exponents for all species falling below ~3 ($P < 3 =1$) (Figure 4.3).

Biomass per unit area estimation

Across all 204 transects conducted in both Queensland and Western Australia, *Catalaphyllia jardinei* had the highest average biomass per unit area ($g \cdot m^{-2}$), followed by *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Micromussa lordhowensis*, *Trachyphyllia geoffroyi*, and *Homophyllia australis* (Table 4.2). As expected, when considering the described exponential size-weight relationship, larger corals contribute disproportionately to coral transect biomass per unit area. This is indicated by the considerable deviation between the sample and population means for *C. jardinei* and *D. axifuga* as shown by standard error values and comparison of mean to median values (Table 4.2). *C. jardinei* in particular had some relatively extreme estimates, with a maximum biomass per unit area value of 33,745.09 $g \cdot m^2$ for one transect resulting from a concentration of large specimens (18 corals > 50 cm), with one coral having a maximum diameter of 90.67 cm.

Table 4.3. The a) average transect biomass per unit area (g·m^2) and b) maximum diameter (cm) of corals recorded on transects, including state that the sample was obtained from, number of transects (n), mean ± *SE*, median, minimum (min) and maximum (max) value, by species and state.

Species	State	n	mean ± *SE*	median	min	max
a) Biomass (g·m^2)						
Catalaphyllia jardinei	QLD	16	6406.00 ± 2985.86	38.00	0.23	33,745.09
Duncanopsammia axifuga	WA	31	102.29 ± 50.45	8.87	0.26	1,462.80
	QLD	2	5.48 ± 5.10	5.48	0.38	10.58
Euphyllia glabrescens	WA	27	17.44 ± 8.08	5.63	0.18	216.62
	QLD	18	60.05 ± 27.69	3.87	0.01	471.65
Homphyllia cf. *australis*	QLD	45	6.01 ± 1.38	3.23	0.27	49.28
Micromussa lordhowensis	QLD	18	10.20 ± 2.61	7.28	0.48	46.58
Trachyphyllia geoffroyi	WA	29	4.31 ± 0.84	2.21	0.10	20.00
	QLD	18	16.28 ± 2.40	4.76	1.69	37.53
b) Diameter (cm)						
Catalaphyllia jardinei	QLD	16	14.10 ± 2.76	8.25	2.79	40.60
Duncanopsammia axifuga	WA	31	10.20 ± 1.01	8.28	4.29	27.70
	QLD	2	8.94 ± 3.77	8.94	5.17	12.70
Euphyllia glabrescens	WA	27	11.70 ± 1.56	10.2	1.00	24.80
	QLD	18	6.37 ± 0.54	5.72	3.02	15.40
Homphyllia cf. *australis*	QLD	45	4.12 ± 0.17	3.80	2.52	7.01
Micromussa lordhowensis	QLD	18	9.29 ± 1.00	8.08	3.50	20.00

Trachyphyllia geoffroyi	WA	29	5.49	± 0.38	5.04	3.76	10.10
	QLD	18	5.04	± 0.23	5.14	1.64	7.41

For species where a sufficient number of transects were available for between-state comparison of biomass (i.e., *Euphyllia glabrecens* and *Trachyphyllia geoffroyi*), only *T. geoffroyi* was found to be significantly different (Bayesian ANOVA pairwise comparison, P=1), with transects from Western Australia on average containing significantly lower biomass per unit area. Similarly, the average diameter of *E. glabrescens* was found to be significantly different (P=1) between regions, with a smaller average size observed for samples from Queensland.

4.4. Discussion

Establishing size-weight relationships for heavily targeted coral species is an important first step towards informing sustainable harvest limits (Pratchett, 2021a). Placing coral harvests into an ecological context is a core requirement for implementing a defensible stock assessment strategy, and this need is particularly critical given escalating disturbances and widespread reports of coral loss (De'ath et al., 2012; Hughes et al., 2017b; Mellin et al., 2019). Using these relationships, managers can now easily sample and calculate biomass per unit area. It is important to point out that all sites sampled in our study represent fished locations, and there is no information available to test whether standing biomass has declined due to sustained coral harvesting at these locations. While these data may now provide a critical baseline for assessing the future effects of ongoing fishing, it is also important to sample at comparable locations where fishing is not permitted or has not occurred (where possible), to test for potential effects of recent and historical harvesting.

Biomass per unit area data presented herein highlights the highly patchy abundance and biomass of targeted coral species (Bruckner and Borneman, 2006), which is evident based on the often vastly different mean and median values (Table 4.2). Examining biomass per unit area estimates for *C. jardinei* for example, which returned some of the highest biomass estimates, the 33.75 $kg{\cdot}m^{-2}$ maximum estimate from a transect stands as an extreme outlier, with 12 of the 16 other transects being below 0.2 $kg{\cdot}m^{-2}$. This indicates the challenges of managing species that occur in patchily distributed concentrations, particularly in a management area the size of the QCF. It is also important to note, these estimates are generated only on transects where the target species occurred, and therefore, should technically not be considered as an overall estimate of standing biomass. While the estimation of size-weight relationships is a step towards a standing biomass estimate, many challenges remain in terms of sampling or reliably predicting the occurrence of these patchily distributed species. Bruckner and Borneman (2006) attempted to overcome this management challenge in a major coral fishery region of Indonesia by categorising and sampling corals (in terms of coral numbers) in defined habitat types, and then extrapolating to estimated habitat area based on visual surveys and available data. This approach, utilising size-weight relationship derived biomass per unit area estimates (instead of coral numbers), may be a viable method for the QCF, however much more information is needed to understand the habitat associations (e.g., nearshore to offshore), and environmental gradients that influence the size and abundance of individual corals. Fundamentally, it is also clear that much more data is required to effectively assess the standing biomass of aquarium corals in the very large area of operation available to Australian coral fisheries.

These corals are found in a range of environments, and it is important to consider available information on life history if attempting to use coral size-weight relationships to inform management strategies via standing biomass estimation. All corals in this study can be found as free living corals (at least post-settlement) in soft-sediment, inter-reefal habitats, from which they are typically harvested by commercial collectors (Pratchett, 2021a). However, only four of the 6 species are colonial (*C. jardinei, D. axifuga, E. glabrescens, M. lordhowensis*) while the remaining two species (*H.* cf. *australis* and *T. geoffroyi*) are more typically monostomatous or solitary. As indicated in previous work (Chapter 3; Pacey et al., 2022), if larger colonial corals were to be fragmented during harvesting instead of removed entirely, fishery impacts would likely be lessened (Chapter 3; Pacey et al., 2022). Given the power relationship between coral maximum diameter and weight, larger corals contribute disproportionately to the total available biomass of each species in a given area. The potential environmental benefit of leaving larger colonies (at least partially) intact is not limited to impacts on standing biomass, as this practice would likely be demographically beneficial given the greater reproductive potential (i.e., fecundity) of larger colonies, which also do not need to overcome barriers to replenishment of populations associated with new recruits (i.e., high mortality during and post-settlement (Penin et al., 2010). This conclusion was drawn largely from data on branching taxa (e.g., *Acropora*), which are relatively resilient to fragmentation and commonly undergo fragmentation as a result of natural processes (Lirman, 2000; Smith and Hughes, 1994; Wallace, 1985). *D. axifuga* can be considered to exhibit a relatively similar branching growth form, however, the growth form of *E. glabrescens* and *C. jardine*i changes with size, moving from small discrete polyps to large phaceloid and flabello-meandroid colonies, respectively (Pratchett, 2021a). While larger colonies of *E. glabrescens* and *C. jardinei* may be relatively resilient to harvesting via fragmentation, the same may not be true for smaller colonies, or species with massive growth forms such as *M.*

lordhowensis. Typically, for each species, the average reported weight was quite low, coinciding with the lower end of the sampled maximum diameter range. For colonial species, the harvested smaller maximum diameters (if fragments) are ideal from an ecological perspective as this will have the least impact possible on standing biomass, and may also leave a potentially mature breeding colony intact. Ultimately, in light of these considerations, the development of uniform and standardised industry-wide harvest guidelines to balance economic and ecological outcomes may be necessary. The development of these guidelines would require consultation with commercial harvesters, as well as considerable additional work in measuring ecological impacts and better understanding the cost of these impacts from an economic perspective. Conversely, if whole colonies are collected, which is necessarily the case for solitary species such as *H.* cf. *australis* and *T. geoffroyi* (and potentially smaller colonies of other species such as *E. glabrescens* and *C. jardinei*); smaller colonies may be collected before they reach sexual maturity, hindering their ability to contribute to population replenishment. Therefore, collection of small fragments should be encouraged for colonial species; while for monostomatous species where this is not possible, introduction of a minimum harvest size based on sexual maturity should be considered.

Additionally, the need for further consideration of the selectivity of ornamental coral harvest fisheries (Dee et al., 2014; Harriott, 2003; Harriott, 2001) when assessing standing biomass is evident. Due to various desirable traits, the majority of available biomass may not be targeted by collectors. As emphasised in this study, the focus on smaller corals is indicative of the trend towards collection of most of these species at the lower portion of their size range, at least compared to some of the maximum sizes recorded on transects (e.g., see Table 4.1 & 4.2 section b). However, it is also important to consider that transects were conducted in areas subject to commercial collection and are likely to skew results and prevent clear conclusions relating to size selectivity. Sampling of unfished populations (i.e., any residing outside of permitted fishing zones) and/or spatial and temporal matching of catch data and transect data across a larger sample of operators will be required to properly address industry size selectivity trends. For instance, only 17.5% of *C. jardinei* corals measured on transects fell within the diameter range represented by data obtained from collectors, with 81.9% of corals measured on transects exceeding this range. If it is viable to collect fragments from larger colonies (which does appear to be the case for some corals such as *C. jardinei*), then a larger proportion of standing biomass outside of this size range could be targeted by fishers. As an additional consideration, only desirable colour morphs of these corals will be harvested, and due to lack of appropriate data, the prevalence of these morphs remains unclear. *H.* cf. *australis* and *M. lordhowensis* for example often occur in brown colour morphs, which are far less popular in markets where certain aesthetic qualities (e.g., specific, eye-catching colours or combinations of colours) are desired, such as the ornamental aquarium industry. Even without delving into further considerations such as heritability of phenotypic traits, management conclusions drawn from standing biomass estimates may be ineffective in the absence of efforts to account for selectivity in this fishery.

The relationship between size and weight was found to differ between all corals, with the exception of *C. jardinei* and *E. glabrescens*. There can be some moderate similarity in skeletal structure between these two species, particularly between small colonies, reflecting the similar maximum diameter range of sampling in the current study. Subsequently, inherent physiological constraints may be imposed on corals that prevent the maintenance of growth rates between corals of smaller and larger sizes, for example, as the surface area to volume ratio declines with growth (Dornelas et al., 2017). In the current study, all corals, with the exception of *C. jardinei*, showed evidence of allometric growth, as exhibited by an estimated exponent value different to 3. Sample size for *C. jardinei* was greatly limited, as this species typically forms extensive beds, and are rarely brought to facilities as whole colonies.

Therefore, the lack of evidence for allometric growth may reflect higher error for the species coefficient parameter due to the comparatively small sample size for this species. This suggests that mass would not increase consistently with changes in colony size in 3 dimensions (Dornelas et al., 2017), which seems likely considering the change in exhibited form described for *E. glabrescens* and *C. jardinei* previously. In the current context, this indicates that the estimated 'a' and 'b' constants are likely to vary as the sample range increases, reflecting the changes in the size-weight relationship between smaller and larger samples of these species. Therefore, ideally, these models should incorporate data that reflect the maximum diameter range of the species in the region of application to allow increased accuracy of biomass estimation. To achieve this will require additional fishery-independent sampling, as large colonies are rarely collected whole, though may be collected as fragments depending on the species. Sampling may be challenging for some species given the difficulty of physically collecting and replacing large whole colonies, particularly for inter-reefal species such as *M. lordhowensis*, which can occur in deep, soft sediment habitat, subject to strong currents. Importantly, obtaining *ex situ* or *in situ* growth rate data should be considered a priority for the management of heavily targeted species. This data is likely to be another necessary component (in conjunction with size-weight relationships) of any stock assessment model developed for LPS corals, and may also eliminate the need to collect large sample colonies to improve estimated size-weight relationships.

The disproportionate focus on smaller corals (i.e., corals in the current study averaged between 4.28 – 11.48 cm in maximum diameter) is likely to lead to an underestimation of weight in corals at greater diameters when used as inputs for size-weight models. This may explain the apparent minor underestimation observed in some species (e.g., *M. micromussa*, *T. geoffroyi*). In the current context, this represents an added level of conservatism with estimates obtained from these equations. While the relationship between size and weight was particularly strong for some species, (mainly *D. axifuga* and *T. geoffroyi*), for other species, such as *M. lordhowensis*, growth curves tended towards underestimation at larger diameter values. As the mass of a coral is reflective of the amount of carbonate skeleton that has been deposited (Pratchett et al., 2015), the coral skeleton may increase disproportionately to coral diameter if or when corals start growing vertically. For example, in massive corals such as *M. lordhowensis*, vertical growth (i.e., skeletal thickening) is often very negligible among smaller colonies, with thickening of the coral skeleton only becoming apparent once the coral has reached a threshold size in terms of horizontal planar area. Additional fisheries-independent sampling outside of the relatively narrow size range of harvested colonies will be required to address this source of error in future applications. Ecological context in the form of fishery independent data on stock size and structure is essential for effective management, especially in ensuring that exploitation levels are sustainable and appropriate limits are in place. Coral harvest fisheries offer managers an ecologically and biologically unique challenge, as the implementation of standard fisheries management techniques and frameworks is hampered by their coloniality and unique biology, as well as a general lack of relevant data for assessing standing biomass and population turnover, not to mention the evolving taxonomy of scleractinian corals (Cowman et al., 2020). Similarly, fishery-related management challenges such as the extreme selectivity in terms of targeted size-ranges and colour-morphs, plus the potentially vast difference in the impact of various collection strategies (i.e., whole colony collection vs fragmentation during collection) also complicates the application of typical fisheries stock assessment frameworks. The relationships and equations established in the current work offer an important first step for coral fisheries globally by laying the groundwork for a defensible, ecologically sound management strategy through estimation of standing biomass, thus bridging the gap between weight-based quotas and potential environmental impacts of ongoing harvesting. It is important to note that the species selected for the current work do not represent the extent of heavily targeted LPS corals. For example, *Fimbriaphyllia ancora* (Veron & Pichon, 1980), *Fimbriaphyllia paraancora* (Veron, 1990), *Cycloseris cyclolites* (Lamark, 1815), and *Acanthophyllia deshayesiana* (Michelin, 1850) are examples of other heavily targeted corals of potential environmental concern (Pratchett, 2021a), and management would also benefit from the estimation of size-weight relationships for these species. Moving forward, the next challenge for the coral harvest fisheries will be to comprehensively document and track the standing biomass of heavily targeted and highly vulnerable coral stocks, explicitly accounting for fisheries effects and also non-fisheries threats, especially global climate change.

Chapter 5

Considerations for the development of a quantitative stock assessment framework for *Acropora* corals in the Queensland Coral Fishery

5.1. Introduction

Australia's coral harvest fisheries are traditionally considered to be well managed (Harriott, 2001; Rhyne et al., 2012; Wood et al., 2012). The Queensland Coral Fishery (QCF) is the largest of Australia's state managed fisheries, both in total allowable commercial catch (TACC; 200 t) and in geographic extent, spanning permitted zones of the 344,000 km^2 Great Barrier Reef Marine Park (Pratchett, 2021a). In contrast to historic perception of Australian coral harvest fisheries, market access for the Queensland aquarium coral industry was recently (December, 2021) restricted following a targeted import ban from the United Kingdom on the grounds of 'non-detriment' concerns (European Commission, 2023; Gay, 2022). Early the following year (January, 2022), the 27 member countries of the European Union also followed suit on an import ban for five major aquarium coral species, this time affecting all Australian fisheries (European Commission, 2023; Gay, 2022). While the UK has since (January, 2023) lifted the restriction on 4 Queensland corals species (European Commission, 2023; Gay, 2022), the import ban of Queensland coral to the EU and UK reflects both accelerating harvest and heightened expectations for rigorously defensible harvest strategies and limits. Specifically, the European CITES scientific review group highlight the major unmitigated growth in Australian coral exports post-2018, as well as the lack of recent substantiating data to provide supporting evidence to implicate harvest has not detrimentally affected wild populations of targeted species (European Commission, 2023; Gay, 2022). It seems inevitable that the accelerating decline of coral reefs globally, combined with public perception of the existential threat posed to these systems by anthropogenic actions, will culminate in growing environmental, public, and political pressure for all for extractive industries utilising coral reefs. Operating against this backdrop, the need to establish quantitative management methodologies to deliver rigorous and data-based assessment of target species is transitioning from a desirable goal to one necessary for the continued operation of wild harvest ornamental coral (and similar) fisheries globally.

Developing appropriate management regimes for the rapidly evolving QCF has been a constant challenge over the last 2-3 decades as harvest has transitioned from focus on ornamental curios, into primarily live rock, and then towards live aquarium corals. Coral harvesting has been regulated in Queensland since 1932. In the 1980s, assessments were undertaken by Oliver (1985) and Oliver and McGinnity (1985), who reported that 86% of coral harvested was used for souvenirs, 10% as dead corals in aquaria, and only 4% as corals for live aquaria (Harriott, 2001). During this time, Oliver and McGinnity (1985) simply assumed that harvest must be below maximum sustainable yield as less than 1% of reefs were open to fishing. The majority of harvest during this period was focused on fast growing 'small polyp stony' (SPS) Pociloporidae coral taxa, in particular, *Pocilopora* cf. *damicornis* and *Acropora* species. From the 1980s into the late 1990s, improvements in aquarium technology led to a drastic shift away from curio souvenirs towards 'live rock' and live coral pieces, primarily 'large polyp stony' (LPS) inter-reefal species. Live rock is composed of coral skeleton pieces (typically with growth of encrusting reef biota such as coralline algae) and can be used to form the vertical substrate within a saltwater aquarium. As of 1999, only 30% of harvested corals collected were dead ornamental corals for use as crafts, curios, souvenirs, and aquarium decorations (i.e., not live rock or live coral; Harriott, 2001). At this time, the spatial scale of the QCF management strategy was significantly less than the contemporary area open to commercial harvesting. Management was structured around 50 authorised coral collection areas, a total allowable commercial catch (TACC) of 4 tonnes per year within each designated zone, for a total of 212 tonnes per year. The average area of a designated zone being 0.025 km^2, the estimated total harvest area would be roughly 1.25 km^2 of the Great Barrier Reef Marine Park. Fishers at the time exploited only 25% of available TACC (Harriott, 2001) across the 50 designated harvest zones, which represents 50 available harvest licenses. It is on the basis of harvest within these designated zones that that the idea of 'less than 1% of total coral biomass is harvested' was first developed (Harriott, 2001), a supposition that has endured within the industry and related literature long past its relevance. Harriot (2001) went on to use estimates of calcium carbonate ($CaCO_3$) accretion during the last inter-glacial period (i.e., last 8000 year period) at One Tree Island (Marshall and Davies, 1984) to further extrapolate that the assumed 2,500 reefs of the great barrier reef accumulate more that 5 million tonnes of $CaCO_3$ per year, of which (at the time) only ~50 tonnes was harvested annually. While little else was available to provide estimates, using the fossil record of calcium carbonate accretion is not appropriate to estimate accumulating standing biomass of living corals, and the relevance to current coral species and environmental conditions in the Anthropocene is highly questionable given the unprecedented rapid and accelerating change in climactic conditions and the impacts of these changed conditions in coral reef systems (Hughes et al., 2017b). During this period, justification for harvest limits and assessment of potential risk has typically relied on such estimates, with additional assumptions and extrapolations drawn from the very limited information on replenishment potential for targeted taxa (i.e., coral growth and recruitment; Harriott, 2001); a practice that would not satisfy present non-detriment requirements. The designated zone approach was not favoured by fishers, who argued that the zones did not intersect sufficiently with populations of LPS corals and were in favour of a roving license approach. Though the Great Barrier Reef Marine Park Authority was reportedly initially opposed to roving licenses on the basis that it would greatly reduce enforcement and impact monitoring capabilities, this management approach was introduced as of 1st July 2006 coinciding with the contemporary 200 t TACC across permitted zones of the Great Barrier Reef Marine Park split into 30% 'specialty coral' and 70% 'other coral' (Pratchett, 2021a).

Due to a lack of *in situ* fishery independent data, contemporary harvest limits for the QCF have been determined according to catch history (Pratchett, 2021a). 'Catch only' assessment methods are commonly used in data-deficient and small scale fisheries (Nielsen et al., 2018; Walsh et al., 2018). If not combined with precautionary limits or other harvest control rules, these methods have a tendency to result in a higher probability of overfishing, as well as higher number of overfished populations (Ovando et al., 2022; Walsh et al., 2018). Put simply, without quantitatively establishing the environmental impact of harvest in the first instance, managing harvest based only on previous catch can result in applying an already detrimental harvest level as a non-detriment or 'pristine state' baseline (e.g., see Garrabou et al., 2017). The QCF has a total harvest limit of 200 t, split between 140 t for 'other coral' taxa, typically characterised as fast-growing and branching SPS coral species; and 60 t for 'specialty corals' characterised as slower growing, fleshy, and sometimes solitary LPS coral species. Temporary harvest limits have for the first time been implemented at the species/genus level in the QCF following the periodic reassessment (Pratchett, 2021a, 2021b) and conditional approval (Commonwealth of Australia, 2021) as a Wildlife Trade Operation. These measures limit current annual catch to 19.5 t. In contrast to catch only management strategies in the absence of additional harvest control rules, which are prone to higher overfishing risks; the precautionary implementation of harvest control rules can lead to underestimation of maximum sustainable yield (MSY) to combat the perceived but unquantified risk of over-exploitation (Walsh et al., 2018). Harvest limits were introduced for 10 species and one genus by applying a multiplier to a historic catch reference period based on level of concern (6 low concern, multiplier of 1.0; 5 moderate concern, multiplier of 0.8). A baseline restriction was also established for all other species to constrain continued unmitigated harvest increase below a level of 150% of the historic reference period. To illustrate, the SPS coral genus *Acropora* had the highest harvest level of any moderate or low concern taxa at 105,977 pieces. This genus is considered to be of moderate concern due to the volume of harvest, steep increase in harvest levels, and lack of taxonomic resolution surrounding both harvest and the group in general. As a result, (following necessary piece to weight conversion) a weight-based harvest limit of 19.5 t was applied, a stark contrast to the potential (though never actuated) prior harvest limit of 140 t. Additionally, this restriction can be lifted only following a ‘rigorous stock assessment’ of the group, precautionarily freezing the harvest limit of this genus until quantitative evidence of non-detriment can be provided (Commonwealth of Australia, 2021; Pratchett, 2021a).

A stock assessment model is a long established management tool that can form a major foundational element of a quantitative management strategy for living natural resources (Fletcher, 1978; Fox Jr, 1970; Graham, 1935; Gulland, 1983; Pella and Tomlinson, 1969; Ricker, 1975; Schaefer, 1957, 1954), which has seen considerable development and is used widely in in many different forms and fisheries currently (Maunder and Punt, 2013; Nielsen et al., 2018; Smith and Addison, 2003). Management frameworks foundationally based upon mathematical stock assessment models represent a considerable advantage to managers and may offer a solution for meeting the ever-increasing expectations of non-detriment finding for key target coral taxa (Maunder and Punt, 2013; Nielsen et al., 2018; Ovando et al., 2022). Furthermore, management strategies informed by ecological data are often vital in industries such as fisheries where these natural resources are composed of populations of individual living organisms that are subject to fluctuation resulting from both fishery and environmental interaction (Maunder and Punt, 2013). Ecologically relevant stock assessment frameworks built on robust probabilistic models represent a desirable scenario for managers responsible for understanding, quantifying, and mitigating the environmental risk of living natural resource harvest (Link and Browman, 2014; Marshall et al., 2019). This would represent an ideal management scenario for an industry such as aquarium coral harvest, where a taxonomically diverse set of organisms (with highly varied biology and life-strategy) are directly extracted from an ecologically delicate, complex, and already anthropogenically threatened environment. However, while an integrated ecosystem based stock assessment framework may be a desirable strategy for managers, there are many scales or levels over which this approach is applied, and it is often an asymptotic goal; approached, but never fully actualised (Link and Browman, 2014; Möllmann et al., 2014). The development of strategies is often slow; limited by timeframes and available funds, and without an assessment framework, determining the resources required to establish quantitative strategies can be difficult (Marshall et al., 2019). Managers can find themselves in a closed loop, whereby it is difficult to establish a stock assessment framework without insight into required knowledge and data deficiencies; while simultaneously, without a stock assessment framework from which to work already in place, it is difficult to determine how and where to establish new data inputs and knowledge gap bridging exercises.

Acropora corals are a major target taxa of particular ecological significance in the aquarium coral trade and within the QCF (Pratchett, 2021a; Wood et al., 2012). This genus was the predominant group in catch composition accounting for just over 32% of harvest in the 2019-2020 financial year, eclipsing other coral taxa, the next most significant contributor (*Homophyllia* cf. *australis*) representing just over 8% (Pratchett, 2021a). The increase in harvest has also been considerable for this genus, jumping from just under 83,000 pieces (just under 1.9 t) in the 2016-2017 financial year, to just over 138,000 pieces (an estimated 3.2 t) in the 2019-2020 financial year (Pratchett, 2021a). Though this genus is highly abundant, harvest is highly selective and only corals with specific phenotypic characteristics are targeted (Pratchett et al., 2020a); in essence, greatly diminishing the potential pool of harvestable biomass. The major concern for this genus is the potential cumulative impact of harvest combined with non-fisheries impacts (Pratchett, 2021a). *Acropora* corals are of considerable ecological significance as a major reef-building taxa that often disproportionally contribute to the structure of contiguous coral reefs (Trapon et al., 2011). The taxonomy of these species remains unresolved (Bridge et al., 2023), and often species are organised into 7 recognised, though further collapsible, growth forms (i.e., ‘bottlebrush’, ‘corymbose’, ‘caespitose’, ‘digitate’, ‘table’ and ‘staghorn’; Veron, 2000). Though this is a purely phenotypic (as opposed to genetic) organisation, *Acropora* growth forms tend to differ in terms of biological and ecological traits and interactions (Chapter 3; Madin and Connolly, 2006; Pacey et al., 2022; Zawada et al., 2019). *Acropora* corals are particularly susceptible to drivers of stochastic change, some of which (i.e., mass coral bleaching events, severe tropical storms and cyclones) are expected to increase in intensity, frequency, or both as a result of anthropogenic influence (Cheal et al., 2017; Hughes et al., 2017b; Kang and Elsner, 2015; Sobel et al., 2014). *Acropora* corals can be disproportionately impacted following thermal coral bleaching events, with post-bleaching mortality resulting in declines of 75% for tabular and staghorn forms on Indo-Pacific reefs (Hughes et al., 2018b), and even up to 95% reported on the Northern Great Barrier Reef (i..e, Lizard Island; Wismer et al., 2019). Some *Acropora* growth forms are more susceptible than others to dislodgement, fragmentation, and mortality because of their specific hydrodynamic properties (Madin, 2005; Madin and Connolly, 2006). Notably, *Acropora* corals are colonial and can reproduce asexually through fragmentation, meaning that dislodgement, fragmentation, as well as disease and predation may result in only partial mortality (Dornelas et al., 2017; Madin et al., 2020). *Acropora* (particularly staghorn and table growth forms) are also favoured by coralivorous invertebrates such as Crown of Thorns Starfish (Keesing, 2021; Keesing et al., 2019) and *Drupella* gastropod snails (Bruckner et al., 2017; Cumming, 2009). Drivers of differing spatial and temporal scales often co-occur and interact, as exemplified by instances of coralivorous invertebrates hindering recovery of *Acropora* coral populations post-bleaching (Bruckner et al., 2017; Keesing et al., 2019). These are just some examples of the complex interacting and co-occurring drivers of *Acropora* population dynamics, and these difficult to disentangle interactions are an inevitability, particularly when considering a system at the scale of the 344,000 km^2 Great Barrier Reef Marine Park.

Given the ecological importance of *Acropora*, the mounting fishery-independent environmental pressure, and accelerating harvest; quantifying the impact of fishing within an ecological (ideally stochastic) and probabilistic context should be a priority for ensuring rigorous evidence of non-detriment. However, attempting to develop a population model for this genus of coral across a geographic scale as large as the Great Barrier Reef Marine Park poses some significant challenges. Firstly, due to the colonial nature of *Acropora*; recommended harvest techniques that reduce potential impact (i.e., fragmentation during collection to leave part of target colony intact; Chapter 3; Pacey et al., 2022); and the weight based TACC system; any stock assessment model aiming to quantitatively bridge the gap between coral harvest and ecological impacts must consider *Acropora* in terms of standing biomass. Secondly, in order to develop an ecologically relevant stock assessment model, key biological stock parameters (i.e., recruitment, growth), as well as stochastic environmental drivers of population dynamics must also be quantified into units that can be incorporated into a stock population model. Multiplying potential pitfalls and tribulations, the taxonomy of this group is unresolved and complex, although it has been demonstrated that growth forms do represent diverging life strategies that interact differently with the environment (Madin and Connolly, 2006; McCowan et al., 2012; Muko et al., 2013; Swierts and Vermeij, 2016), and only particular phenotypes are targeted for harvesting. Advantageously, *Acropora* is one of the most well studied scleractinian coral genera (Ball et al., 2021), with considerable research existing to inform understanding of growth, recruitment, and drivers of stochastic change (Baird et al., 2012; Dornelas et al., 2017; Gold and Palumbi, 2018; Madin et al., 2020; Pratchett et al., 2020c; Souter et al., 2010; Wallace, 1985). Similarly, *Acropora* is usually incorporated in major scientific monitoring surveys (AIMS, 2017), and the Great Barrier Reef (though vast) is also one of the most studied coral reef systems internationally. Even so, it is unclear how much of this existing research, much of which focuses on *Acropora* colonies as singular units subject to partial mortality (e.g., Madin et al., 2020), can be translated into fishery relevant biomass-based units. This work aims to utilise the available data where possible, to develop an initial biomass population model, and through this exercise; examine management actions and data inputs that may be required for the development of a rigorous working population model for *Acropora* corals. Through the development of an initial model, the efficacy of current management strategies and arrangements can for the first time be considered through a stock-model based assessment framework lens. More specifically, this work will explore whether it is at all possible to quantify the risk associated with varying levels of *Acropor*a harvest currently, what are the potential shortcomings in doing so. Furthermore, in light of the historic management scale change, the suitability of the current

QCF management scale will be considered; and whether there is any evidence to suggest scale and/or region is an important consideration when attempting to incorporate ecological input data. As evidenced by recent trade bans (European Commission, 2023; Gay, 2023, 2022), non-detriment requirements will increasingly be expected to be delivered using rigorous, data-based, and ecologically relevant means. This exercise will allow evaluation of the potentiality, as well as foreseeable hurdles, of transitioning wild coral harvest in Australia from a catch-based management arrangement, towards a rigorous, integrated data-based assessment approach.

5.2. Methods

5.2.1. Natural stochasticity

Representation of natural stochasticity via independent ecological survey data (i.e., percent coral cover) forms a central part of the biomass-based population model here described. Percent cover is a common metric used in scientific benthic surveys internationally, whereby the substrate type and occupancy of that defined substrate type is recorded within an outlined survey area (e.g., Emslie et al., 2020; Holbrook et al., 2008; Jokiel et al., 2015; Jones et al., 2022). The Australian Institute of Marine Science Long Term Monitoring Program (AIMS LTMP) is one of the largest and longest running marine survey programs in the world carried out across up to 130 reefs of the Great Barrier Reef (AIMS, 2017). This program includes detailed records of coral cover (%) across permanent transects at a range of taxonomic levels. To make use of available environmental data, it was required that a relationship between *Acropora* coral cover and biomass be established. This was achieved using i) collected size-weight relationship data; ii) processed video transect data; from which both biomass (g) and corresponding coral cover (%) estimates were derived (see Chapter 3; Pacey et al., 2022).

Coral cover for each available transect was estimated by summarising the planar area of corals (i.e., greatest diameter by perpendicular diameter to the nearest cm) measured for each 50 m x 1 m video transect. A frequentist linear model was used (due to the simplicity of the intended application) to describe the relationship between total *Acropora* coral cover (%) and total biomass (g) for each transect within the statistical computing software environment R (ver 4.2.2; R. Core Team, 2022), for both a GBR and Cairns region model, the former utilising all available data and the latter only data within the area defined as the Cairns region (from -18°3’0.0” to -16°3’0.0” latitude and from 143°56’60.1” to 149°37’48.0” longitude; Figure 5.1). R code available via Pacey (2024).

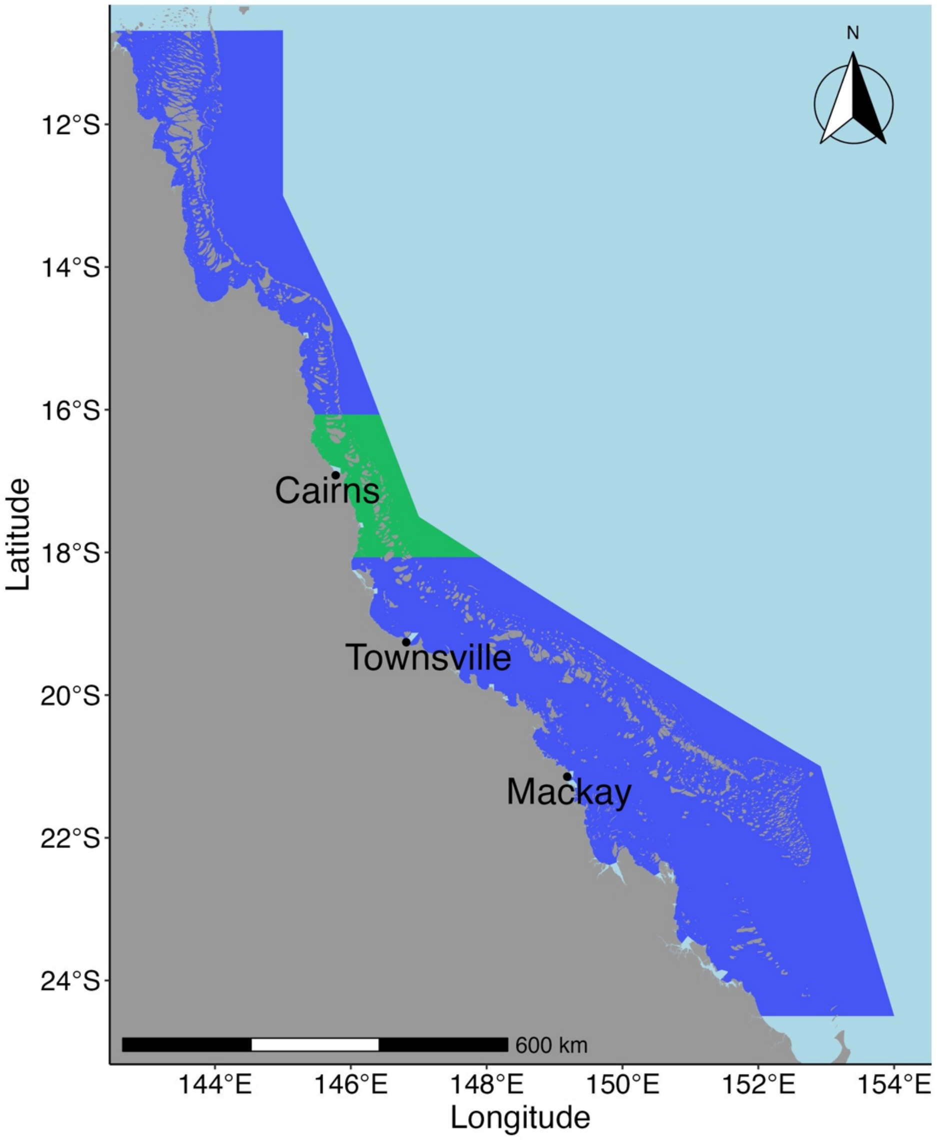

Figure 5.1. Map depicting the spatial bound of the QCF scale model (i.e., whole Great Barrier Reef Marine Park; blue and green area) and the Cairns region scale model (green only). Grey represents mainland and reef polygons.

Once this relationship was established using available data for both the Great Barrier Reef and Cairns region models, AIMS LTMP *Acropora* coral cover (%) measurements (summarised to the transect) were converted to a per transect total biomass (g) estimate. From these estimates, year to year percent change in biomass was characterised. The biomass estimates for each transect were summarised to reef (and then averaged to adjust for survey effort), to allow characterisation of reef level percent change in biomass. To characterise natural change in *Acropora* biomass in the absence of fishing, only data from survey sites excluding commercial coral harvest permitted zones and relevant to geographic bounds defined for each model were selected. Similarly, as population models operate over years iteratively, only instances of continuous year-to-year surveys were included in the calculation of year-to-year biomass change (%) over a contemporary time-period (2005 to 2022). Using change in biomass (%) data meeting these specifications, a Bayesian General Additive Model was used to describe change in *Acropora* biomass (%) over time across reefs for both the GBR and Cairns region model. This allowed the trend in change in biomass (%) to be probabilistically represented over this time frame across reefs meeting selection criteria by sampling from the model posterior expected predictions.

5.2.2. Model structure

The basic underlying model is a slightly modified version of the Schaefer (1955, 1957) biomass dynamic model, being a simplistic recursive function into which three main components are fed; biomass change value (%; '*C*'), initial stock value (t; '*I*'), and carrying capacity (*t*; *P*). Firstly, the model-relevant expected prediction posterior is sampled, pulling 100 consecutive biomass change values (without replacement) to represent the yearly natural stochastic change for the model year ('*t*') range (0 to100). Next, the initial biomass (in tonnes, t) value is calculated. Initial biomass (t; i.e., the start value of model run at $t = 0$) was estimated as below (see Equation 5.1) using two parameters; estimated total reef area (m^2; '*r*') and average biomass per unit area (g · m^{-2}; '*a*'; Table 5.1).

$$I = B_{t=0} = \frac{r \cdot a}{1 \cdot 10^6} \; (1)$$

Estimated total reef area (m^2; 'r') was calculated using the *gisaimsr* package (Barneche and Logan, 2023). The extent of area was summarised from shapefiles representing reefs within park boundaries open to fishing. The process was also repeated for the Cairns region model, incorporating only reefs within the defined geographic bounding box (ymin = -18° 3' 0.0", ymax = - 16° 3' 0.0", xmin = 143° 56' 60.1", xmax = 149° 37' 48.0") constructed using the *nominatimlite* package (Hernangómez, 2023). Average biomass ($g \cdot m^{-2}$; 'a') was estimated according to available data within each region (GBR scale and Cairns region) according to available video transect data (40 transect overall, 18 transect Cairns region), processed to allow an estimate of biomass for each transect (see Chapter 3; Pacey et al., 2022 for more detail).

Carrying capacity (t; 'L') was estimated according to two parameters, total reef area (m^2; 'r') and maximum estimated biomass per unit area ($g \cdot m^{-2}$; 'H'), as described below (Equation 5.2).

$$L = \frac{r \cdot H}{1 \cdot 10^{-6}} \; (2)$$

Similar to estimated biomass, maximum biomass (t; see Table 5.1) was calculated using data outlined in Chapter 3 (Pacey et al., 2022), assigned according to geographic bounds of model (i.e., GBR, Cairns region). For each year (t), within the recursive function itself, the biomass change (proportion) value ('C') is converted back into a natural biomass (g) quantity using the supplied biomass (g) value (t) from the conclusion of the last calculation ('B_{t-1}'). To calculate the biomass ('B') value for year 't', annual catch (t; 'M') must be incorporated, taking the form as below (Equation 5.3).

$$B_t = (1 + C) \cdot B_{t-1} - M \text{ (3)}$$

When ‘B_t’ > ‘L’, ‘B_t’ = ‘L’; when ‘t’ = 0, the value ‘B’ = ‘I’; and when $B_t \leq 0$, $B_t = 0$.

5.2.3. Iteration and replication

To examine the influence of a range of possible values for defined variables (i.e., average biomass ‘a’, maximum biomass ‘H’, total reef area ‘r’, annual catch ‘M’) a reference grid containing a combination of all defined values was established using the range of values defined in Table 1, via the function ‘expand_grid’ from the *tidyr* package (Wickham et al., 2023).

Table 5.1. Range of model parameters for biomass (g·m^2; '*a*'), maximum biomass (g·m^2; '*H'*), total reef area (km^2; **'*r*'**), and annual **catch** (t; '*M*'), included in *Acropora* biomass population model for the Great Barrier Reef region model as well as the Cairns region model. 'Low' indicates lower bound estimate, and 'high' indicates higher estimate bound.

		GBR model	Cairns region model
Biomass (g·m^{-2}) '*r*'·10^{-6}	Minimum	2.70	3.18
	Q1	4.14	4.65
	Median	6.87	10.32
	Mean	8.74	11.37
	Q3	12.82	15.24
Max biomass (g·m^{-2}) '*H'*	Low	12.88	12.88
	Estimated	25.75	25.75
	High	51.51	51.51
Total reef area (t) '*a*'	Low	8672.46	865.19
	Estimated	17344.93	1730.38
	High	34689.85	3460.77
Annual catch (t) '*M*'	Values	0, 20, 200, 2000, 20000, 200000	0, 20, 200, 2000, 20000, 200000

Using in combination the function 'replicate' (Base R), and the 'pmap_dfc' function of the *purrr* package (Wickham and Henry, 2023), each combination of the values in Table 5.1 was supplied to the above defined recursive population model 1000 times. Each run (spanning years 0 to 100) was then be summarised according to whether stock collapse (i.e., stock levels reach or fall below 10% of initial biomass g; i.e., $B_{t=0}$) occurred during the run, allowing an indicative probability of collapse for each combination of variables (e.g., 988 out of 1000 instances of collapse). Considering a fishery collapsed at 10% of initial catch (or in this case, initial estimated population biomass) is a common fisheries management convention (Li and Smith).

5.2.4. Logistic regression

Bayesian logistic regression (logit link) was used to investigate the conditional relationship between supplied fishery parameters (i.e., average biomass, maximum biomass, total reef area, annual catch) and probability of collapse (i.e., *n* instances of collapse out of 1000). Models for the GBR and Cairns region were constructed using the probabilistic programming language for statistical inference *stan* via the r packages *rstan* (Stan Development Team, 2023) and *brms* (Bürkner, 2017a). Model validation was conducted via the packages *DHARMa* (Hartig, 2022) and *rstan*. For a full description of model and parameters, see Appendix D, Table D1.

5.3. Results

5.3.1. Natural stochasticity

There was a significant linear relationship between biomass (g) and coral cover (%), although the parameters of this relationship varied between available data within the defined spatial bounds of the Great Barrier Reef (Linear regression; $F_{(1,38)} = 733.8$; $R^2 = 0.95$; $p < 0.05$) and Cairns (Linear regression; $F_{(1,16)} = 491$; $R^2 = 0.97$; $p < 0.05$) model (Figure 5.2, a & b). The rate of increase was comparatively gradual in the GBR biomass (g) vs coral cover (%) model compared to the Cairns region model (coefficient estimate β = 10,584.7 c.f. β = 15,724.2), while maximum coral cover (%) did not extend over the same sample range as the GBR model. Bayesian GAMs used to characterise observed natural stochasticity in units of biomass percent change in coral cover differed between regions, with the smooth term for the GBR model being minimally wiggly and non-significant (Bayesian GAM smooth term; estimate = 26.61 ± 25.50; lower 95% C.I. = 0.8, upper 95% C.I. = 92.3), while the Cairns model had a significant smooth term (Bayesian GAM smooth term; estimate = 40.15 ± 35.26; lower 95% C.I. = 1.7, upper 95% C.I. = 133.3), although the estimate could still be considered minimally wiggly. While the non-significant smooth term for the GBR model suggests that a linear model could effectively be used instead, for the current application, the danger of over-fitting was considered minimal (in fact, representing the nuance of this relationship would be advantageous in representing trends probabilistically), and so it was decided to continue with the GAM approach to keep methods as uniform as possible between the two regions. Additionally, population level effects were not significant for either the GBR (β estimate = 16.5, lower 95% C.I. = -171.3, upper 95% C.I. = 168.5) or Cairns region model (β estimate = 136.47, lower 95% C.I. = -322.3, upper 95% C.I. = 232.4), indicating that the relationship between coral cover was not a clear linear per-unit increase or decline, as seen in Figure 5.2, c & d. As indicated by the 95 % C.I. values in Figure 5.2 c & d, and confirmed by tails (Change %; GBR, min = -44.4, max = 65.0; Cairns, min = -82.2, max = 137.5) in Figure 5.2 e & f, the Cairns region was subject to greater variation in biomass percent change (%). The centre of the density curve for each region also differed, with the median value for the GBR region being 11.7 % increase (95% HDCI; lower = -7.9, upper = 30.3), while the value for Cairns was higher at 28.3 % increase (95% HDCI; lower = -10.8, upper = 64.1).

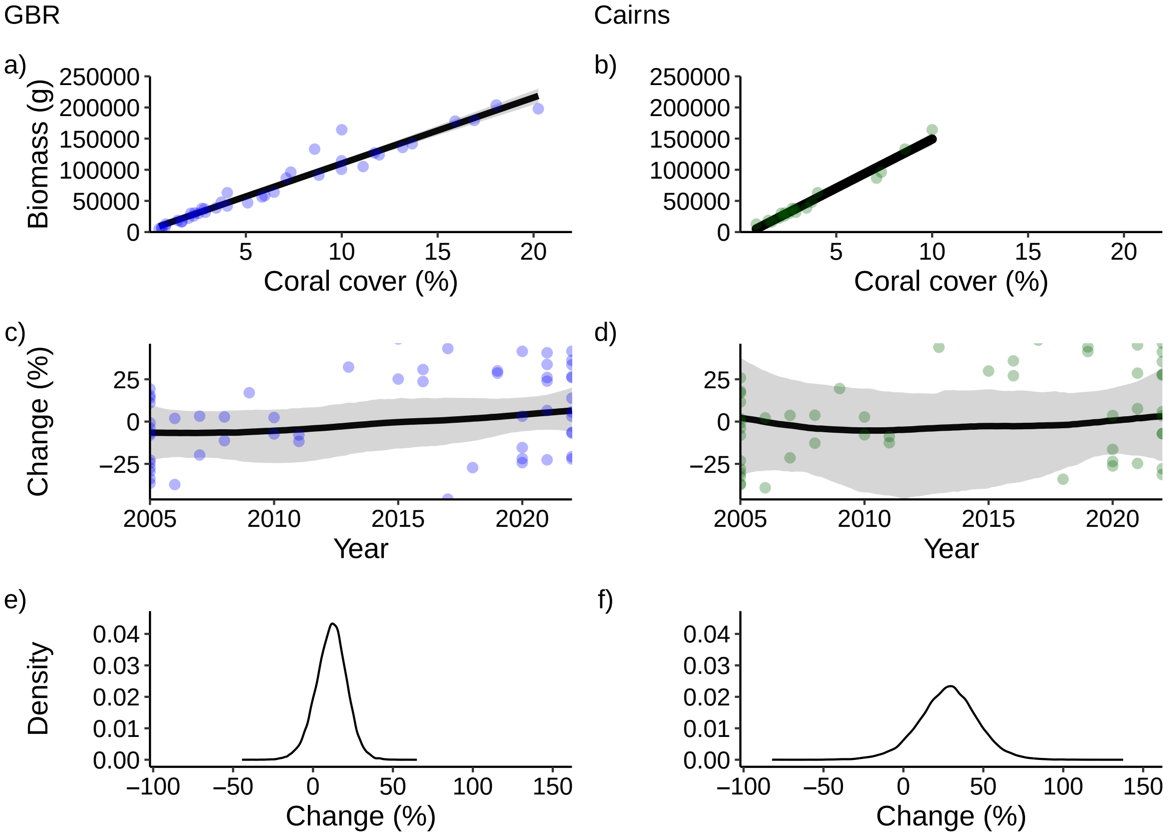

Figure 5.2. Depiction of key underlying relationships for both the Great Barrier Reef (GBR) and Cairns region population models. Panels a) and b) show the relationship between coral cover (%) and estimated total biomass (g) for each transect (black line), with underlying data (coloured data points) originating from data outlined in Chapter 3 (Pacey at al., 2022) for the GBR and Cairns region models respectively. Panels c) and d) show the model line of the Bayesian GAM (black line) and 95% probability band (grey outline) and underlying datapoints, each representing a biomass change (%) value from consecutive sampling years at an AIMS LTMP surveyed reef that meets the zoning and geographic extent requirement for the GBR and Cairns region population models respectively. Panels e) and f) displays the relationship between posterior draw density and biomass change (%) to illustrate the shape of the expected model posterior draws for the GBR and Cairns region respectively.

A total of 270 stock variable combinations were explored (see Table 5.1), repeating each combination 1000 times over a run time of 0 to 100 years, for a total of 270,000 individual runs and 27,270,000 data points (i.e., stock at year value) for all model runs, each for the Great Barrier Reef and Cairns model. Generally, model runs continued to increase until reaching carrying capacity, with multi-step dips below capacity followed by eventual recovery (coloured lines; Figure 5.2). Due to the differing carrying capacities between regions (resulting from region specific estimated reef area), the GBR region has a maximum stock level exceeding 400,000 (t), while the Cairns region maximum stock level is slightly over 40,000 (t). As natural stochasticity was applied in terms of biomass change (i.e., relevant to stock values and parameters within each model), it appears that the Cairns region has higher stochasticity, with more frequent dramatic changes (the potential for which was noted above), as individual model runs (Figure 5.2; green lines) more commonly approached initial stock estimate value compared to the GBR model. Visually, increasing annual catch appears to have negligible effects (although a slight increase in the number of runs with stochastic stock drops can be discerned in the Cairns model), until a threshold is reached, where model runs fail to increase in stock value. Once this threshold is reached (at 2,000 for the Cairns model, and 20,000 for the GBR model), model runs fail to increase substantially, and some runs also drop to 0 (Figure 5.2).

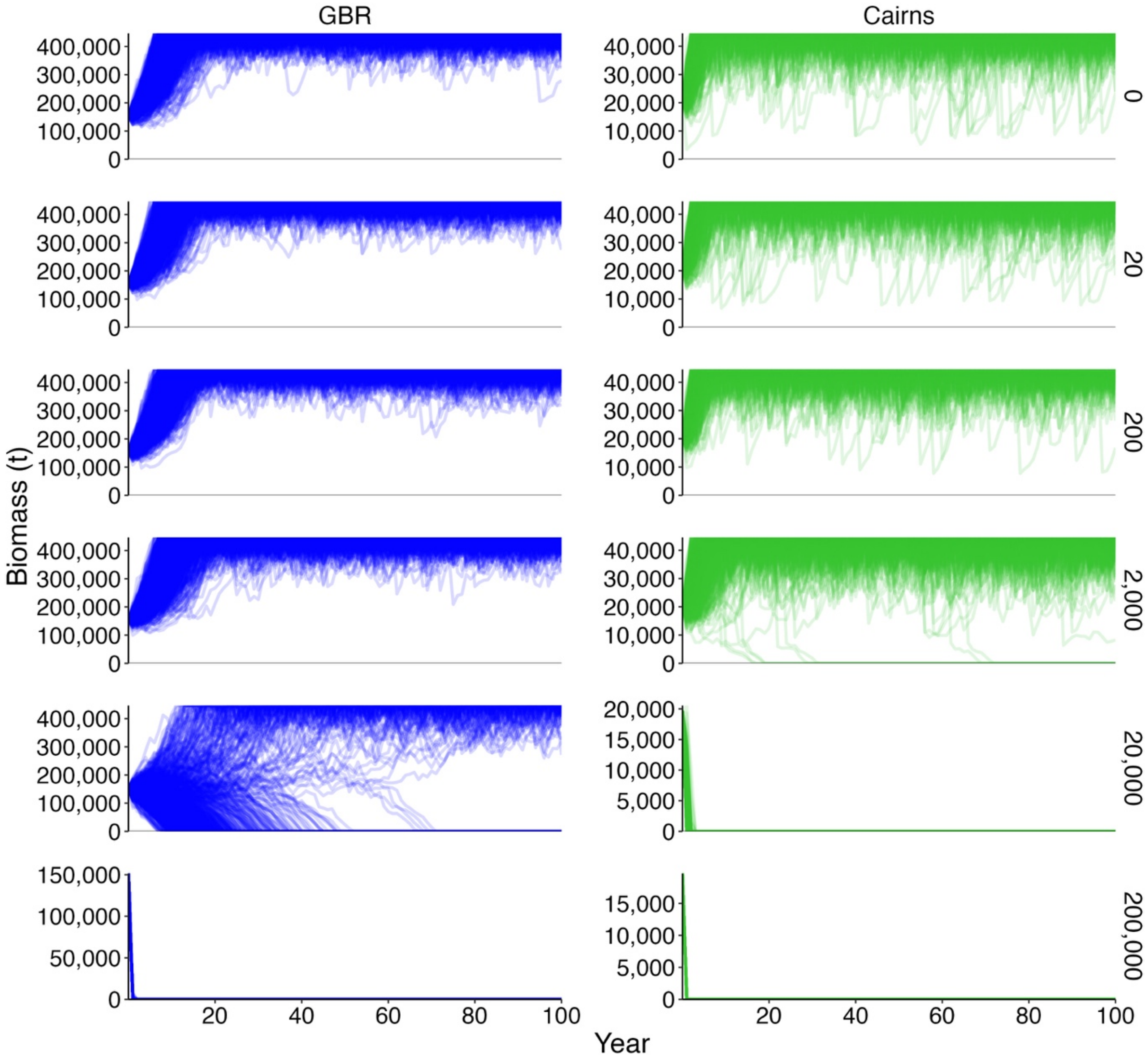

Figure 5.3. Predicted stock (t) across model years for the Great Barrier Reef population model and Cairns region scale population model respectively, showing the relationship between stock (t) and year for each model run (coloured lines; n = 1000) when; biomass (g·m^{-2}) ‘*a*’ = average biomass value, max biomass (g·m^{-2}) ‘*H’* = estimated value, and total reef area (km^{2}) ‘*r*’ = estimated value (see Table 5.1 for values).

5.3.2. Logistic regression

Logistic regression analyses of run outcome probabilities for the GBR model suggested that over the range of supplied variables, all variables (average biomass, max biomass, total reef area, and annual catch) were significantly influential on the probability of fishery collapse, however with the exception of annual catch (t) exploration of conditional effects plots suggested that these relationships are likely non-linear and may require alternative modelling approaches to capture. For GBR annual catch (Bayesian logistic regression; mortality β estimate = 76.0, lower 95% C.I. = 74.5, upper 95% C.I. = 77.6), the probability of fishery collapse in the GBR model was estimated as 1.85×10^{-7} with 0 t annual catch, 1.89×10^{-7} probability of collapse at 19.5 t annual catch, and 2.14×10^{-7} at 140 t. According to this model, LD50 (i.e., 50% probability of collapse) occurs at 14,965.2 t annual catch. Certainty of fishery collapse (P = 1) is estimated to occur at 31,192.0 t of fishing.

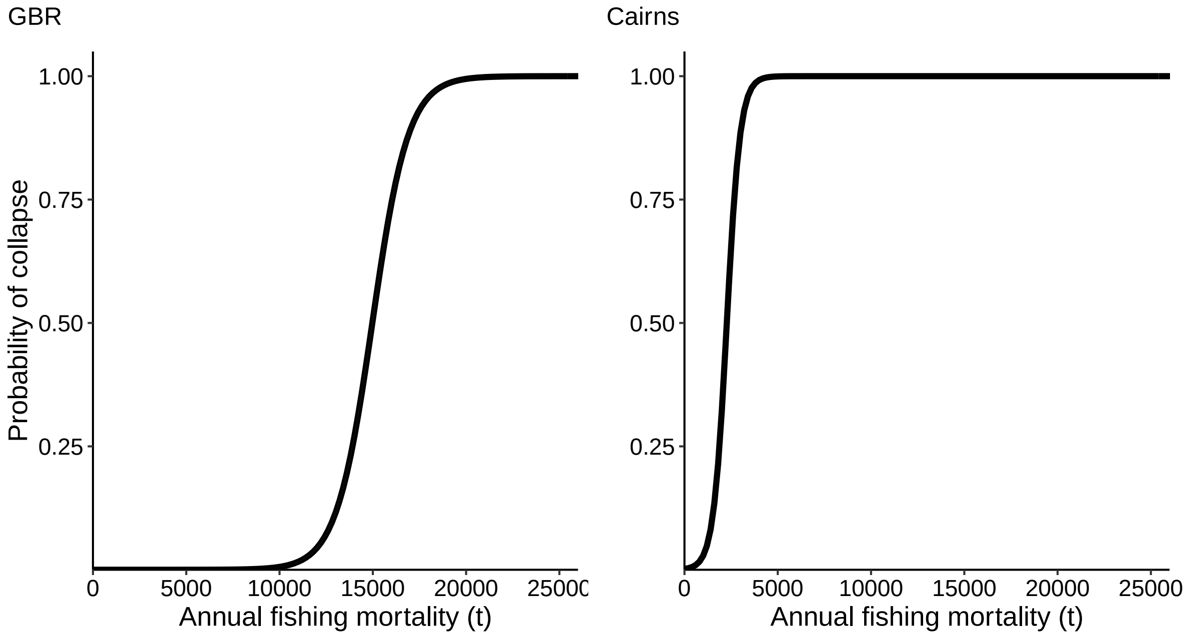

Figure 5.4. Conditional relationship displaying the estimated probability of fishery collapse (i.e., when estimated stock biomass (t) ≤ 0.1 of initial biomass $\boldsymbol{B_{t=0}}$ during model run) and annual catch (t) for Great Barrier Reef (GBR) scale model and Cairns scale model.

Due to convergence issues, model factors with the exception of annual catch were excluded from the Cairns region model, although this again likely indicates that there was not a linear relationship between model parameters and probability of collapse as also observed in the GBR model. Fishery mortality was again a statistically significant influence on collapse probability (Bayesian logistic regression; mortality β estimate = 200.4, lower 95% C.I. = 196.1, upper 95% C.I. = 205.0; Figure 5.3). At 0 t annual catch, the probability of fishery collapse was estimated to be 1.91 x 10^{-3} at 0 t annual catch, 2.01 x 10^{-3} at 19.5 t, and 2.79 x 10^{-3} at 140 t fishery mortality. For the Cairns region model, LD50 occurs at approximately 2,291.9 t, and probability of collapse becomes a certainty at approximately 8,446.8 t of annual catch.

5.4. Discussion

The work here presented moves towards establishing a quantitative integrated management framework for *Acropora*, the most heavily targeted genus by harvest volume and catch composition in the QCF. A relationship was established between total biomass (g) and coral cover (%) for *Acropora* corals. Establishing the existence of a total biomass (g) and coral cover (%) relationship is valuable for the QCF as a fishery that is highly deficient in independent ecological data. While the underlying parameters of this relationship are likely to change according to many factors (i.e., region, species composition, sample location), the existence of this relationship presents means of utilising environmental data with further strategic sampling. Specifically, coral cover (%) measurements from the Australian Institute of Marine Science Long Term Monitoring Program (AIMS, 2017) were converted to total biomass (g) estimates using this relationship. The total biomass (g) estimates generated for reefs meeting the geographic requirements for inclusion (i.e., within GBR or Cairns region and non-fishing permitted zone) allows, through quantification of year-to-year biomass change estimates at the reef level, projection of simulated stochastic change in stock population models across the geographic scale of each model. While also subject to limitations regarding sampling, scale, and the inherent limitations of forward projection based on past trends; establishing this methodology represents a valuable informative step towards ecologically integrated management for the QCF. Finally, following the methodology outlined, it was possible to conduct a quantitative, probabilistic assessment of fishery collapse (i.e., reduction of stock to 10% of initial biomass or less) risk, and while keeping in mind the limitations of the specific results here reported, such a possibility has not previously been realised for the Queensland Coral Fishery. Most importantly, this work highlights key considerations of current management strategy and geographic management scale. The current geographic management scale of the QCF, an area of 344,400 km^2, precludes not only the possibility, but also the necessity of evenly distributing monitoring and impact quantification efforts of this patchily distributed and selective harvest fishery across the extent of total or larger regional spatial scales.

The difference observed in coral cover (%) and biomass (g) relationships between GBR and Carins region scale models exemplifies the underlying challenge associated with developing a robust *Acropora* harvest model across a fishery management area as geographically large as the QCF. The distribution and intensity of sampling required to achieve reasonable representation of relationships to allow extrapolation to areas outside sampling is unclear; but due to logistical limitations, further *in situ* strategies would be a necessity at both model scales presented here. The relationship between *Acropora* coral cover (%) and *Acropora* biomass (g) varied between the Great Barrier Reef region scale model and the Cairns region scale (β coefficient estimate = 10,584.7 c.f. β = 15,724.2), with the coefficient for the Cairns relationship being somewhat steeper. This (at least partially) resulted in a wider posterior between the GBR region scale (min = -44.4, max = 65.0) and the Cairns region scale (min = -82.2, max = 137.5) and so, yearly stochastic change of greater magnitude. There are many factors to consider that may affect the underlying relationship between coral cover (%) and total biomass (g), which complicates the characterisation of relationships, particularly when extrapolating across geographic scales of increasing spatial extent. At fine scale, *Acropora* coral biomass is largely determined by deposition of the calcium carbonate skeleton, a process that is highly plastic and strongly influenced by biotic and abiotic factors (Álvarez-Noriega et al., 2018; Gladfelter, 1984). *Acropora* is also a highly diverse and phenotypically plastic genus (Bay et al., 2009; Bridge et al., 2023). The unresolved state of *Acropora* taxonomy is well known (Ball et al., 2021; Bridge et al., 2023), and on the Great Barrier Reef alone this genus is composed of potentially hundreds of species of six or more recognised growth forms (Veron, 2000). Therefore, the composition of species and growth forms sampled is likely to alter to an unknown extent the biomass (g) and coral cover (%) relationship, as a result of regional differences in species distributions, bathymetry, as well as climactic and weather patterns all have the potential to influence these relationships (Madin, 2005; Oliver et al., 1983). It will be necessary to understand the relevance of these fine scale differences within a fisheries context when attempting to construct rigorous biomass (g) stock assessment frameworks and supporting models. The potential for accumulation of considerable error increases exponentially as fishery independent sampling intensity decreases and geographic area over which these relationships are extrapolated increases, and so further consideration on how to properly account for these biases in sampling will be a key development step.

For a genus such as *Acropora* that is highly reactive to stochastic natural disturbance events (Trapon et al., 2011), the development of an ecologically relevant stock assessment model requires an understanding of natural biomass change in the absence of fishing. Currently, disentangling and quantifying individually the effects of various disturbance events (e.g., mass coral bleaching, cyclones and tropical storms, irruptions of corallivores such as CoTS and *Drupella*), as well as key stock assessment dynamics such as growth, recruitment, and population connectivity in units and at a scale relevant to the QCF does not appear viable due to the temporal and spatial resolution of existing data (despite the extensive spatial and temporal range of this data; AIMS, 2017). These population dynamics occur over a broad range of temporal and spatial scales ranging from metres to thousands of kilometres, and from days to decades (Adjeroud et al., 2016; Ritson-Williams et al., 2009), and knowledge and incorporation of relevant effect scales (e.g., via a disturbance relevant geographic impact scaling mechanism) into population models would benefit realistic representation of biomass change. Although somewhat of a black box in terms of individual stochastic event impact, a simple approach to capture the stochasticity surrounding the amalgam of these dynamics may be the most viable option currently reconcilable with available data that can allow incorporation of natural stochastic change into stock assessment of corals in the QCF. In this study, this approach was taken by converting existing *Acropora* coral cover (%) data to biomass (g), and then developing a probability distribution via Bayesian General Additive Modelling, to project the current patterns in potential stochastic change (%) in biomass (t) forward over a 100 year period. The current projections of *Acropora* cover indicate a probabilistically positive increase (based on GAM model posteriors; posterior median 11.7 % for GBR and 28.3 % for Cairns region) across geographically relevant (to each model) reefs that are closed to fishing. This result is consistent with the general trends outlined in the AIMS 'annual summary report of coral reef condition 2021/2022', which attributed recent recovery of hard coral cover largely to fast growing *Acropora* corals (AIMS, 2022). As well established, this net increase trend is undermined by the susceptibility of this genus to stochastic disturbance (Ortiz et al., 2021; Pratchett et al., 2020c), which is also reflected in the biomass (g) percent change posterior values, with the potentiality of biomass reductions exceeding (i.e., Cairns) or approaching (i.e., GBR) 50% yearly. Given the accelerating degradation of coral reefs globally, particularly as a result of mass thermal coral bleaching (Hughes et al., 2017b), and the possible increase in occurrence and/or severity of other major disturbances (e.g., cyclones) attributed to global climate change (Cheal et al., 2017; Sobel et al., 2014), the biomass change (%) may result in a reduction (or at least a year to year shift) of this median positive increase. Therefore, it is important that stochastic biomass change (%) trends are periodically updated to reflect most recent available data.

Current results concerning the probability of stock biomass (t) collapse (occurring at or below 10% of initial biomass) indicate that while probability of collapse was in each case lower at the current 19.5 t catch limit (GBR: 1.89×10^{-7} and Cairns: 2.01×10^{-3}) compared to previous (potential) catch limit of 140 t (GBR: 2.14×10^{-7} and Cairns: 2.79×10^{-3}); the probability of stock biomass collapse was still miniscule, and comparable to the probability at 0 t annual catch (GBR: 1.85×10^{-7} and Cairns: 1.91×10^{-3}). The probability of fishery collapse was considerably higher in the Cairns region scale model compared to the Great Barrier Reef scale. This is likely a result of the interplay between the smaller geographic extent of available reef area (~9% that of the GBR scale model), as well as the greater stochastic biomass change (%) range in this region (Figure 5.2, e & f). Obviously, the same amount of fishing over a smaller geographic area will result in a higher probability of collapse in that geographic range. These results also suggest the need to consider risk based on the region in which harvesting occurs. However, there are several reasons (aside from the obvious sampling and data limitations) that the level of risk associated with an annual catch of 200 t is deceptively low in both models. Current taxonomic resolution of the genus *Acropora* within the scientific literature, and within fishery data collection, is not sufficient to represent the potential risk of local extirpation to heavily targeted *Acropora* species/phenotypes. Coral harvest fisheries for the aquarium coral trade target very specific *Acropora* phenotypes (e.g., the ‘strawberry shortcake’ variation of *Acropora* c.f. *microclados*), most of which do not have resolved taxonomy, and may be composed of multiple species with distinct ranges (Bridge et al., 2023). Therefore, while the genus of *Acropora* is highly widespread, there is a distinct lack of environmental data pertaining to the distribution of heavily targeted phenotypes, as well as a lack of knowledge about the recruitment potential of these species. It is unclear at what geographic scale it is relevant to consider coral recruitment within a fisheries context, and in reality, there is unlikely to be any clear answer, as relevance is likely to change with scale, region, and the species in question according to the prevailing hydrographic, climatic, and weather conditions. However, this uncertainty is compounded for *Acropora* corals, as without a clear understanding of the factors determining the desired phenotype (i.e., genetic and environmental). While it is clear recruitment will depend on myriad factors, such as the distribution and abundance of the phenotype in question, the bathymetry, and the genetic heritability of the desired phenotypic traits. Without knowledge of the recruitment potential of a specific phenotype (and without being able to properly identify the taxonomy/taxonomies of this species within an area subject to fishing), it is potentially problematic to assume replenishment from potentially genetically distinct or distant population will refresh harvested populations. Attempting to answer these questions at a scale of the Great Barrier Reef Marine Park management area is currently logistically impossible, and a highly impractical approach.

If attempting to crystalise the general attitude towards coral harvest expressed in early QCF management, it was typically posited that the sheer size of the Great Barrier Reef Marine Park and the assumed biomass of *Acropora* available for harvest must by necessity mean that fishery impacts are inconsequential (Harriott, 2001). However, the current lack of capacity to detect fishery impacts should not be equated with a legitimate lack of impact, or indeed an inconsequential probability of detrimental effect. One alternative management approach that could drastically increase impact quantification would be simply to focus on developing frameworks in smaller, discrete geographic areas based around the distribution of fishery effort for key aquarium coral species, such as *Acropora*. Unquestionably, the biomass of *Acropora* corals accessible to fishers eclipses current and historic quotas and reported harvest levels (Chapter 3; Pacey et al., 2022). In practice, fishers are bound to very distinct geographic areas of operation as a result of onshore facility locations and logistical limitations; from which there are typically harvested a very distinct set of species and phenotypes. If management areas were to be based on smaller scale distribution of fishery effort, instead of being faced with the daunting task of attempting to represent trends across a management area the size of the Great Barrier Reef Marine Park; data collection at a representative and harvest appropriate scale becomes (at least comparatively) feasible, or at least the effort required can be properly estimated using the spatial effort data as a reference. While spatial data outlining fishing effort is not readily available to the public currently, this data is recorded as part of the fisher logbook exercise carried out by all commercial harvesters in the QCF, with catch recorded to a geographic resolution of 6 x 6 nm square grids. These 6 x 6 nm grids could form the basis for the deployment of much needed fishery independent field sampling to establish the parameters included in the initial population model outlined in this work. Adjoining grids with concentrated fishing for the same defined phenotype (or species) can be amalgamated into one management area. This would also allow prioritisation of resources as these zones are established based on target species and fishing effort. Due to the unknowns surrounding recruitment, the most conservative approach would be to treat non-adjoining grid amalgamations as separate management entities, requiring their own set of sampling and utilising a distinct model incorporating ecological data. As hinted, for this approach to be effective, further taxonomic resolution is required in data recorded by the fishery. While the unresolved nature of *Acropora* taxonomy is a complication, all that is really required in the immediate future is a fishery relevant catalogue of targeted *Acropora* corals. While this would require a great deal of further research, it is an achievable goal. Similarly, adjacent areas that are not subject to fishing would need to be included as a reference point, and to build understanding of natural processes in (or at least as close as possible) to defined regions. While it would likely be possible to use existing park zoning to this end, incorporating harvesting-free reference zones within the newly defined management area bounds is another option. These zones would only need to be observed by commercial coral harvesters to be effective, and so would not impact other users.

A grid-based management approach would also benefit other key fisheries species, and the underlying population model applied over these management units could easily be modified to reflect their biology and ecology. Many heavily targeted, or potentially higher risk species, are harvested from turbid soft-sediment inter-reefal environments where they are typically concentrated into patchy aggregations. Unlike *Acropora*, some of these key species are single polyp, solitary species, examples being *Trachyphyllia geoffroyi* and *Homophyllia australis*. Solitary species are generally considered to be slow-growing, and over-exploitation has traditionally been viewed as the major threat (Bruckner and Borneman, 2006; Rhyne et al., 2014). A more traditional age (or in this case, size) structured population modelling approach would likely be most suitable for these species. This would require establishing biological parameters pertaining to size-at-maturity, growth, and recruitment *in situ*. Again, this poses an almost insurmountable hurdle to accomplish over the entire management area, but if handled piece-meal via a grid amalgamation approach; though still requiring considerable work, this is at least a more approachable goal. In such instances, it may be important to establish a minimum size limit based on size-at-maturity (or the LD50 of size-at-maturity) to prevent removal of individuals from the population that have not had the opportunity to reproduce. A maximum size can also be set to protect individuals that contribute disproportionality to replenishing the population through higher fecundity associated with larger size. This hints at the individual-centred nature of such models (as opposed to the biomass-based approach used for *Acropora*), and indeed, it may be most appropriate to convert an individual based model to biomass to link to the biomass-based quota. Overall, this population modelling strategy contrasts with the approach here taken for *Acropora*, where it is assumed that the major driver of stock population change is likely to be environmental stochasticity (natural or anthropogenically influenced), and it is most appropriate to model the population in terms of biomass. While this does not equate to inter-reefal species being immune to disturbance events (Pratchett et al., 2020b), fishery over-exploitation should be the foremost concern for these species. It is perhaps best to consider key coral species as falling somewhere on a continuum between the extreme attributes of fast-growing, colonial, branching and reef building corals such as *Acropora*, and the slower growing solitary, inter-reefal species such as *T. geoffroyi*. Establishing the precedent for these two extremes will allow development and adjustment of population modelling strategy for all life-strategies falling in between the branching and colonial c.f. solitary archetypes.

Considerable work is required to continue the development of a population stock assessment model for *Acropora* corals in the QCF. The results here presented are illustrative only of the possibilities and potential of currently resolvable population modelling approaches and are not sufficiently robust to inform current management quantitively, due to limited data inputs to support underlying relationships (e.g., coral cover % and total biomass g) as well as major limitations relating to current understanding of taxonomy and distribution of targeted phenotypes. Furthermore, the initial modelling approach outlined is intended only as a start point to begin developing a robust quantitative and probability-based population assessment framework with the limited fishery relevant data and knowledge currently available. All underlying parameters (e.g., average biomass, maximum biomass, and total reef area) and relationships (e.g., coral cover % and total biomass g relationship) require location and scale specific sampling before these values can begin to be considered near representative. For example, data used to construct the coral cover (%) biomass (g) relationship is geographically limited to the northern half of the Great Barrier Reef Marine Park, with no representation of the southern section. Similarly, the method used to estimate available reef area for the purposes of initial stock estimate and carrying capacity would need to be refined. The use of satellite imagery to estimate reef composition across the defined assessment area would be more appropriate. This would allow the estimation of contiguous reef area excluding other features such as sand banks and seagrass beds, which cannot be delineated from the shapefiles used in the current iteration which lack any fine scale benthic composition data. Estimating available area is likely to represent a considerable challenge for inter-reefal species, as suitable habitat may not be as clearly discernible from satellite imagery. Instead, a more viable approach for such species may be to compile ecological data from considerable survey efforts with satellite imagery, bathymetric, and environmental variables to develop predictive species distributions models.

Quantitatively managing wild coral harvest is a challenging prospect. The ornamental aquarium industry is complex and rapidly expanding. While subject to constant boom and bust cycles, and ever-changing fads, continued and increasing demand for quality and affordable corals seems certain. There is an effort by some fishers to move away from a primarily wild caught to an aquaculture-based business model, however for the foreseeable future, continued reliance on wild caught coral also seems to be an inevitability for this industry. Therefore, considering the pervasive degradation of coral reef environments globally, heightened expectations for justifiable and defensible management strategies will be placed on these industries. Here we examined how an initial quantitative stock assessment model could be constructed using only available data and knowledge. As evidenced, the current management scale of the QCF poses a significant hinderance to managing a highly selective multi-species fishery. The scale examined will almost certainly change and underly any biological/ecological relationships, and the difficulty and risk of mischaracterisation increases exponentially as the geographic management scale also increases. Similarly, the initial population models presented in this work highlight quantitively why the predominating philosophy has been to consider harvest (particularly of SPS species) to pose little consequence due to the sheer scale of the Great Barrier Reef. While risks may in fact be minor, the fact remains that the resolution (ranging from scale to taxonomic) available from current information is insufficient to properly quantify the potential risk harvest poses to targeted coral species. A focus on managing this fishery as smaller discrete geographic units would allow the problem to be approach sequentially, filling in knowledge gaps according to supposed risk. Areas with the highest concentration of fishing for most at risk species can be targeted for vital fishery-independent sampling to collate the ecological and biological data needed to properly characterise the relationships underpinning stock biomass level and change. While considerable work will be required to establish quantitative fishery management for the QCF, Australia is uniquely placed to lead other coral harvesting nations in the continued sustainable management of a rapidly growing internationally industry.

Chapter 6

General Discussion

There is a clearly discernible need for improved management in the Queensland Coral Fishery and other major Australian commercial ornamental coral fisheries, as well as coral harvest fisheries globally. In the QCF and Australia, there exists immense potential for informed, quantitative, ecologically relevant management (Harriott, 2003; Harriott, 2001); especially given established state and federal governmental management mechanisms for Australian (export focused) fisheries. The substantial growth in coral harvest under cotemporary management strategies is certainly not due to an absence of necessary oversight. Rather, established management frameworks and mechanisms have not yet been 'tailored' (i.e., adapted and optimised) to deal with the unique challenges of managing commercial aquarium coral harvest in Australia. In 2006-07 when contemporary management strategies came into effect, up to the 2019-20 financial year assessed during wildlife trade operation approval (Commonwealth of Australia, 2021), there has been a >750% increase in the number of coral pieces harvested, and a 186% increase in harvested coral weight (Chapter 2). Perhaps the most notable changes to come into effect at the initiation of this contemporary harvest period is the shift from a discrete fishing area approach (Harriott, 2001), to a roving license approach (Pratchett, 2021a), combined with a new total annual quota of 200 t split between 70% SPS 'other coral' and 30% LPS 'specialty coral' (DEH, 2006). Despite this substantial increase in reported harvest levels, which met and exceeded triggers for detailed management reviews (Chapter 2; Pratchett, 2021a), there have only very recently been necessary changes in fisheries management arrangements (Commonwealth of Australia, 2021; Pratchett, 2021a).

6.1. Changing management arrangements for Australian coral fisheries

In 2021, all major Australian coral harvest fisheries (i.e., for the states QLD, WA, and NT) were subject to the periodic re-assessment (Pratchett, 2021a, 2021c, 2021d) conducted on behalf of the Australian CITES scientific authority and overseen by the Australian Commonwealth Government (Commonwealth of Australia, 2021). This process is required to reinstate approval of these industries for international export. This latest reassessment of the QCF (Pratchett, 2021a) highlighted a number of worrying harvest trends and outcomes, as reported in Chapter 2. While permission to export was reinstated for the QCF, this permission was granted subject to 9 conditions and two recommendations, acknowledging of the need for substantial revisions to current management arrangements. Aside from species-specific harvest limits that were established based on catch history (Chapter 2), and a default limit of 600 kg implemented for all other CITES-listed corals (Commonwealth of Australia, 2021); these conditions/ recommendations included improvements in species level identification and reporting capacity for corals currently identified to genus, including spatially relevant species harvest composition for *Acropora* spp. (Commonwealth of Australia, 2021). Similarly, the need to establish quantitative assessment informed by *in situ* data was noted in the most recent wildlife trade operation re-assessment (Commonwealth of Australia, 2021; Pratchett, 2021a), and later cited by the CITES scientific review group as a key reason for the recommended trade restriction on Australian corals (European Commission, 2023; Gay, 2022). In the 2021-2026 Queensland Coral Fishery harvest strategy (DAF, 2021), the Queensland Department of Agriculture and Fisheries cites its desire to move towards a biomass-based management approach.

The large area of operation for the QCF has offered a false sense of security to fishers and managers regarding the available resources within its bounds. An attitude towards marine resources as unfathomably ‘boundless’ has historically been an obstacle for sustainable management, with this idea predating the very concept of fisheries management itself

(Jentoft, 2003; Laws, 1997; Nielsen, 1999). This attitude is highly pervasive within fisheries globally, including other coral harvest fisheries (Ferse et al., 2012). The lack of ecologically informed, quantitative justification for the overall harvest limits currently and historically seemingly exemplifies this outlook. The development and longevity of this attitude within the QCF is more understandable within the historical context of harvest (i.e., prior to establishment of the 'roving license within permitted park zones' approach in 2006), at which time harvest was substantially less, and the geographic extent of harvest was conserved to specific bounded areas (Harriott, 2001). However, with the increase in harvest outlined during the most recent Wildlife Trade Operation assessment (Commonwealth of Australia, 2021; Pratchett, 2021a), it is clear that the lack of concern as a result of the 'sheer scale' of coral resources (if ever applicable) has long outlasted its relevant to this industry. This apparent attitude has likely hindered the much-needed development of management within Australia's coral harvest fisheries.

The absence of capacity to detect, quantify, and track impacts at a harvest relevant scale under which these industries have been operating for the last few decades, cannot reasonably be used as evidence for a lack of harvest impact. In other words, if the scale is in fact so vast as to warrant little concern; then there is still a clear need for this to be quantitively demonstrated. However, as implicated by the body of work presented within this thesis, the reality of managing coral harvest and developing quantitative harvest limits, is subject to much greater complexity due to various (and often dynamic) biological, ecological, social, and economic factors. The lack of supporting data to inform prescribed harvest limits has already impacted Australia's prior well-held reputation for leading wild harvest coral fishery sustainability among the international community, with drastic ramifications for market accessibility for Queensland and Australian commercial ornamental aquarium coral fishers (European Commission, 2023; Gay, 2023, 2022). Without action to address these shortcomings, it is reasonable to suppose that the longevity of this industry within Australia is at risk. The ability to estimate standing coral biomass is a crucial step to allow quantification of fishery impacts in a harvest relevant context.

6.2. Status and trends in wild coral stocks

Establishing a methodology to estimate coral biomass *in situ* via video transect using size-weight relationships (Chapter 3 and 4; Pacey et al., 2023, 2022), if utilised, represents a significant step towards improving the data necessary to undermine sustainable management of Australian coral fisheries. These relationships were established for 6 *Acropora* growth forms (bottlebrush, caespitose, corymbose, digitate, staghorn, table; Chapter 3; Pacey et al., 2022) as well as 6 heavily targeted LPS coral taxa (*Catalaphyllia jardinei, Duncanopsammia axifuga, Euphyllia glabrescens, Homophyllia australis, Micromussa lordhowensis*, and *Trachyphyllia geoffroyi*; Chapter 4; Pacey et al., 2023). The utility of these relationships was then demonstrated by estimating the standing biomass of *Acropora* for a single contiguous coral reef (Chapter 3; Pacey et al., 2022), and later used to develop a working biomass estimate based on reef area open to fishing for contiguous reefs to facilitate the Great Barrier Reef scale and Cairns region scale *Acropora* stock biomass models utilised in Chapter 5. The total biomass of Stucco Reef (~4.7 km^2) was estimated to be ~106 t for all *Acropora* growth forms combined (Chapter 3), while the total biomass of *Acropora* corals on the GBR was (for the purpose of modelling) estimated as 151,594.7 t (Chapter 5). The closest comparable estimate was made by Harriott (2001) using a study of geological accumulation of calcium carbonate ($CaCO_2$) to approximate production of all coral taxa on One Tree Island (~1,875 t $\cdot yr^{-1}$). By then extrapolating to the 2,500 reefs of the Great Barrier Reef, this approach yielded an estimate of 5,000,000 t $\cdot yr^{-1}$. Although the purpose of Chapter 5 was to investigate methodology to generate biomass models using only the limited available information, the median change in biomass per year was an 11.7% increase across the GBR, which equates to

17,736.6 t $\cdot$yr^{-1}. This represented approximately 0.4% of the estimate made by Harriott (2001), though Harriott's (2001) estimate was intended to represent all coral taxa. When considering *Acropora* at the genus level, even by these values, total annual harvest (TACC) of 140 t is likely to represent <0.1% of available biomass; although this is a far sight to achieving reliable, quantitative, estimates (as discussed further in Chapter 3, 4, and 5). Additionally, whether consideration of harvest impacts at the genus level across the geographic extent of the GBR offers any legitimate insight appears increasingly unlikely (Chapter 2 & 5; Pratchett, 2021a).

Estimating biomass via extrapolation over habitat area will be comparatively complicated for LPS taxa if attempting to mirror the methodology utilised so far for SPS corals (Chapter 3 & 5). An estimate is currently possible using available information, though the results remain problematic. Harris et al. (2013) estimated submerged habitat area of the Great Barrier Reef, which could broadly be considered suitable for inter-reefal taxa. Within the depth range of harvest fishers (max ~35 m) the extent of potential habitat amounted to approximately 23,827 km^2. Using the average biomass $\cdot$ m^{-2} for *Trachyphyllia geoffroyi* (16.26 g $\cdot$ m^{-2}; Chapter 4) as an example, extrapolating over potential habitat area would result in a total standing stock of 387.9 t for this species. According to this estimate, current harvest annually (1.3 t; Chapter 2) represents 0.32% of potential standing stock. However, the biomass per unit area for this (and other LPS) species is almost certainly a gross overestimate, as sampling was conducted at high concentration aggregations of this taxon, and the otherwise highly patchy distribution is driven by not fully elucidated environmental factors. That is, it is unlikely the extent of area estimated by Harris et al. (2013) is a reasonable representation of suitable habitat for this taxon, and habitat cannot be estimated without additional *in situ* sampling to determine abiotic and biotic factors influencing distribution. Therefore, as described here, current attempts to estimate stock of select LPS corals across the immense geographic scale of the

QCF with available information will almost certainly result in overestimation and should not be granted any credibility until further *in situ* data can be obtained.

The sheer geographic scale of a commercial aquarium coral harvest fishery the extent of the QCF presents a quintessentially Australian problem. While establishing size-weight methodology is a major step towards achieving a quantitative biomass-based management goal, there is still considerable work to conduct in the hopes of reaching a robust and defensible standing biomass estimate suitable for incorporation into a biomass stock assessment model. Specifically, there is a need to determine the appropriate level of sampling effort required to generate a reasonable representation of the biomass of a coral reef (relevant to *Acropora* spp.), or alternatively, of an aggregation of inter-reefal LPS taxa. As indicated, while the considerations required to establish a robust standing biomass estimate will have core similarities, the approach will diverge between SPS and LPS taxa according to life-strategy and ecology.

Consideration must be given to the appropriate sampling scale required for a robust representation of biomass across various geographic extents. The data required to generate an estimate of standing biomass is made up of two main data inputs: those being 1) the size-weight data (i.e., data outlining the planar diameter and respective weight of a coral colony), and then 2) the video transect data (data containing planar measurements of each relevant coral along a 50 x 1 m video transect). While the work presented in Chapters 3 & 4 (Pacey et al., 2023, 2022), describe size-weight relationships for 6 Acropora growth forms and 6 heavily targeted LPS taxa; he skeletal density and/or porosity of coral may be influenced by environmental factors (e.g., location Lough and Barnes, 1992; or depth Ng et al., 2019), meaning that size-weight relationships may also differ across locations, depths, and reef positions. However, from a practical standpoint, the sampling required to determine size-weight relationships is highly intensive *in situ*, particularly for larger specimens (Chapter 3; Pacey et al., 2022) and in the case of the present work (Chapter 3 and 4; Pacey et al., 2023, 2022) sample sizes attained (*n* = 3972 for *Acropora* corals, *n* = 2548 LPS species) were only possible through collaboration with coral harvesters. Therefore, while further work to examine inter-species (or growth form in the case of *Acropora*) variation in size-weight relationships across depths, regions, and positions would be ideal (and of greater importance for *Acropora* in particular due to representation of multiple species), it is unlikely that similar sampling efforts with required additional data can be repeated in the near future; particularly as sampling effort focused on surveying target taxa distribution and population parameters is likely to be more useful for achieving management recommendations (Pratchett, 2021a).

An additional consideration is the sampling effort required, and over what scale, to robustly generate a representative estimate of biomass for a contiguous reef (in the case of *Acropora*) or for a soft sediment LPS species aggregation from video transect data inputs. Starting at the smallest scale, biomass can be reliably estimated over the transect scale using size weight relationships. For SPS *Acropora* coral at the scale of a reef, a sampling effort approach mirroring that of existing monitoring surveys (AIMS, 2017; Emslie et al., 2020) is likely an appropriate start point, although further work will be required to verify this does provide a reasonable representation. The Australian Institute of Marine Science Long Term Monitoring Program (AIMS, 2017; Emslie et al., 2020) utilises 5 permanent 50 m transects at 6-9 m depth contours per reef to characterise coral cover and benthic composition, as well as fish species abundance and diversity, with the 5 sites distributed around the outside perimeter of the sampled reef. If distributed around the circumference of the reef so as to characterise protected and exposed sides, this may provide a useable estimate of average coral biomass $\cdot m^{-2}$ for contiguous reef branching taxa such as SPS *Acropora*, allowing estimation sampled reef biomass.

The next consideration is how the sampling effort required relates to the overarching management scale (i.e., total fishery extent), as discussed in Chapter 5. Bruckner and Borneman (2006) used between 4-6 30 m transects per location across 12 locations (total *n* between 48 to 72) to estimate abundance and size distribution for 8 harvested taxa (~25 species) in the Spermond Achipeligo, Indonesia. This level of effort was used to extrapolate measures across 7 defined habitat types (determined by sight, maps, and spatial data) in each of the 12 fished locations, representing an approximate area of 8,680 m^2. This represents a minimum survey effort of 1 transect to 180.8 km^2. To replicate this survey effort across the Great Barrier Reef Marine Park would require approximately 630 transects (344,400 km^2 · 0.33 percent of park closed to fishing divided by 180.8), although even this represents a very low transect to area ratio that is not relevant to a system as patchily distributed and dispersed along such considerable latitudinal and longitudinal distances. Further, survey effort will most likely need to be replicated for each target taxa. Sampling for SPS taxa such as *Acropora* can be focused around contiguous reefs. LPS taxa can co-occur in in soft-sediment aggregations (Pratchett, 2021a), however, attempts to share sampling effort between taxa may not provide a sufficient estimate of distribution as aggregations are not always multi-species, often being species specific. For LPS taxa, to extrapolate estimation over a GBR or regional scale to the same effect, repeated *in situ* sampling to determine biomass · m^{-2} and parameters driving occurrence may be required in order to construct species predictive models (Elith and Leathwick, 2009; Raven, 2002). Capacity to detect suitable habitat in the same way as contiguous reef (i.e., satellite imagery), as well as the occurrence of taxa within this habitat is considerably limited in comparison due to the depth and turbidity of inter-reefal LPS taxa aggregations, although; a key consideration in this approach is also the availability, resolution, and relevance of existing GBR-scale environmental and bathymetric data facilitating prediction (Harris et al., 2013).

6.3. Spatially discrete monitoring, if not management

It will be very difficult to detect impacts of coral harvesting across the vast area (344,400 km^2) of the Great Barrier Reef Marine Park. However, challenges associated with detecting impacts cannot continue to be conflated with presumed lack of impacts at this large scale. Interestingly, during early discussion of alternative management plans, loss or reduction of monitoring capacity resulting from a roving license approach was reportedly feared by the GBRMPA prior to implementation of contemporary harvest strategies in 2006 (Harriott, 2001). In agreeance with the prediction, the results of Chapter 5 strongly suggests that the transition from a fixed area license strategy to a roving license model (Chapter 1; Chapter 2;Pratchett, 2021a; DEH, 2006) has greatly hindered capacity to monitor harvest at a relevant (and achievable) local scale. Naturally, it may be assumed that simply dividing management area into modules of smaller geographic extent (e.g., Cairns region model cf. GBR scale model; Chapter 5) may overcome this issue. However, while smaller geographic areas may enhance capacity to detect impacts mathematically, simply by virtue of the reduced available habitat in which to project impacts (e.g., GBR vs Cairns region model, Chapter 5); in reality, smaller management areas do not fix the major underlying issue, which is the capacity to implement scale appropriate surveying to acquire crucial ecological ($g \cdot m^{-2}$ estimates, size structure, species distribution parameters) and biological (growth, recruitment, mortality, partial mortality) information.

In situ sampling will be required, not just to develop a quantitative management framework, but to maintain the operation of coral harvest fisheries in Australia long term. Lack of supporting *in situ* data was a key reason cited by the CITES Europe scientific review group for the trade bans enacted against Queensland by the UK, and Australia by the EU (European Commission, 2023; Gay, 2022). It is obvious that the potential effort and monetary investment (i.e., consider the sampling effort and monetary cost of operation for the AIMS

LTMP) required to conduct appropriate sampling at GBR or even region scale makes a blanket approach to monitoring not only untenable, but impractical. However, the spatial catch data required for justification of prioritisation and monitoring of particular areas and species already exists in the form of 6x6 nm grids as recorded by fishers as part of standard logbook reporting (Pratchett, 2021a). Priority should be given to establishing monitoring according to current harvest level, increase in harvest level, spatial concentration (and temporal accumulation) of harvest, as well as any other information/data (e.g., Ecological Risk Assessment) to inform species level risk. For example, priority should be given to the 9 LPS species (*Acanthophyllia deshayesiana*, *Catalaphyllia jardinei*, *Cycloseris cyclolites*, *Duncanopsammia axifuga*, *Euphyllia glabrescens*, *Fimbriaphyllia ancora*, *Homophyllia* cf. *australis*, *Micromussa* cf. *lordhowensis*, *Trachyphyllia geoffroyi*) currently placed under species level restriction due to unconstrained growth (Chapter 2; Pratchett, 2021a). The solitary species *Trachyphyllia geoffroyi* and *Homophyllia* cf. *australis* in particular should receive priority in terms of *in situ* data collection due to the harvest level (1.3 t and 1.8 t respectively), rate of increase (884.8% and 3,074% in retained pieces over 14 year period), and maximum spatial distribution of harvest (maximum >15% and >10% of harvest from single 6x6 nm block; Chapter 2; Pratchett, 2021a). As monostomatous corals, available knowledge (Pratchett, 2021a; Pratchett et al., 2020a) suggests that *T. geoffroyi* and *H.*cf. *australis* are biologically among the most vulnerable to localised depletion, while also being among the most economically important to the trade. Similarly, the annual harvest, increase, and especially the spatial distribution of harvest (Chapter 2; Pratchett, 2021a) suggest the colonial species *Micromussa lordhowensis* should be a high priority. For the major target genus *Acropora*, additional steps are required. The *Acropora* spp. category recorded the largest contribution to catch composition (32%) at over 140,000 pieces in 2019-20 financial year (Chapter 2; Pratchett, 2021a). Therefore, impetus should be placed on establishing a working species level taxonomy in areas of concentrated harvest to facilitate *in situ* monitoring and fishery harvest recording (Chapter 5). This recommendation was also a condition (i.e., condition 5) for Wildlife Trade Operation approval (Commonwealth of Australia, 2021). This would greatly aid development of capacity for the assessment of (at least quasi-)species level local depletion risk. A shift towards 'harvest-intensity-based' management to prioritise species of concern and develop monitoring around areas of spatially concentrated harvest is one method to capitalise on existing spatial harvest data; and allow evaluation of the monitoring costs required to sustainably manage the QCF.

Further to the idea of tailoring management strategies for the QCF, several key harvesting techniques were identified in Chapter 3 and 4, which could be implemented as standard practices in heavily fished areas to minimise environmental impacts. As size-weight relationships were described by 2 factor power relationships for all growth forms/species, larger individuals contribute disproportionately to standing biomass. Larger individuals are therefore of greater value to stock biomass even without accounting for other factors likely to increase with size, such as survival and fecundity (Chapter 3; Pacey et al., 2022). For fast-growing colonial branching taxa such as *Acropora*, fragmentation *in situ* should be encouraged, so as to leave colonies at least partially intact, there-by reducing the removal of stock biomass associated with harvest of an entire large colony. For solitary LPS taxa (e.g., *Acanthophyllia deshayesiana, Trachphyllia geoffroyi, Homophyllia* cf. *australis*), the establishment of minimum and maximum harvest size (a common fisheries management measure; Gwinn et al., 2015; Punt, 2010) may reduce the probability of local depletion as a result of harvest (Chapter 4; Pacey et al., 2023). While size-weight relationships for LPS taxa were also effectively described by a two-factor power relationship (Chapter 4; Pacey et al., 2023), it is not realistic or effective to attempt fragmentation of monostomatous corals *in situ* during collection (Pratchett et al., 2020a). Although further data on minimum size at maturity (but see Pratchett et al., 2020a) would be required, this common fisheries management measure (Gwinn et al., 2015; Punt, 2010) would prevent individuals from being removed before contributing to recruitment. Similarly, individuals that cross a certain (to be determined) value threshold in terms of weight, could also be conserved due to their disproportionate contribution to standing stock (Pacey et al., 2023). Though just a few examples arising from existing research, standardised operational practices represent significant potential to tailor harvest relevant management strategies, and so reduce the potential for local depletion.

6.4. Conclusions and future priorities

Table 6.1. Recommendations for managers, according to taxonomic group.

Targeted taxon group	Recommendation
Overall	1. Implement strategy to provide quantitative biological/ecological justification for prescribed harvest quotas. 2. Consider execution of management strategies in discrete, spatially reduced harvest intensity based units. 3. Prioritise management units for quantitative management strategy establishment according to spatial harvest intensity.
SPS *Acropora* corals	1. Establish a fisheries relevant, spatially explicit taxonomy. 2. Implement strategy to develop quantitative, ecosystem-based management. 3. Consider implementation of specifically defined harvest practices (e.g., surrounding whole colony vs fragment collection), including harvest-size ceiling (i.e., upper limit).
LPS corals	1. Conduct necessary *in situ* surveys to aid in the establishment of standing biomass estimates. 2. Consider implementation of a minimum and maximum harvest size for vulnerable (e.g., monostomatous) taxa.

If properly utilised, this work has the potential to result in marked improvements in the management capacity for ornamental corals targeted for commercial harvest. Chapter 1 provided relevant insight into the international ornamental aquarium coral industry, and the place Australian coral harvest occupies globally, to the extent possible with available data and information. Chapter 2 described contemporary harvest trends in Australia's major coral harvest fishery (i.e., the QCF), underlining the concerning growth in harvest at various taxonomic groupings (total, family, genus/species), and highlighting a number of heavily targeted taxa that have been placed under taxon level restriction. In Chapter 3, methodology to estimate *in situ* biomass of corals using size-weight relationships was described, then utilised to generate size-weight relationships for 6 growth forms of *Acropora* across two distinct bioregions of the QCF using data gathered in conjunction with fishers. In Chapter 4, size-weight relationship were developed for six species of heavily targeted LPS taxa, including two monostomatous species. With both Acropora and LPS taxa, the nature of the two-factor power relationship between size and weight had implications pertaining to sustainable harvesting practices, although due to biology, these practices differed between SPS and (particularly single polyp) LPS taxa. In Chapter 5, the first example of a quantitative stock assessment modelling strategy for *Acropora* corals in the QCF was described and explored, most importantly highlighting the current shortcomings and future data-input considerations. Overall, this work has highlighted the need to reconsider the management arrangement appropriate for SPS corals (Chapter 2, 3, and 5), and highlighted concerns for LPS corals (Chapter 2 and 4; Bruckner and Borneman, 2006; Harriott, 2003; Knittweis and Wolff, 2010). For SPS corals, and particularly *Acropora*, a lack of capacity to detect localised depletion (Chapter 5) should not be equated with limited or negligible risk of over-fishing, even given very large estimates of standing biomass for the genus. The most immediate management recommendation for this taxon is to establish a fisheries relevant taxonomy, to facilitate the recording and measurement of (ideally spatial) species level data. While unconstrained growth is of concern for LPS target taxa (particularly considering the annual harvest level; Chapter 2; Pratchett, 2021a) concentration of harvest is of particular concern, and in situ sampling is desperately needed to assess the current state and impacts for these heavily targeted taxa.

Until now, the QCF has been a victim of its own vast spatial scale. Managing a highly selective, ornamental fishery over such a large spatial scale has greatly reduced capacity to detect fishery impacts, as well as made the investment required to establish the monitoring capability appear seemingly unachievable considering assumed resources and current (or lack thereof) *in situ* monitoring effort (Chapter 5; Pratchett, 2021a). However, detecting harvest impacts is both viable and necessary, because harvesting is spatially concentrated (Chapter 2; Pratchett, 2021and the data on spatial distribution of fishing is already held by Queensland Fisheries. While the spatial concentration of harvest should be a concern to managers, it also provides an opportunity to overcome the illusion of scale and (justifiably) assign key monitoring resources. Considerable effort will be required to sustainably manage these fisheries into the future as coral reefs continue to degrade globally as a result of anthropogenic influence (Hughes et al., 2017a). Though relevant to all coral taxa, for some such as SPS *Acropora* corals, the incorporation of ecosystem-based management will be a necessity for sustainable harvest (Chapter 5). Failure to generate a quantitative management strategy to bring harvest operations into line with global expectations of modern fisheries management (Gulland, 1983; Marshall et al., 2019; Nielsen et al., 2018) will not continue to be tolerated by the global community (European Commission, 2023; , 2022).

Over two decades ago Harriot (2003) published a now seminal work in the field asking the simple question ‘Can corals be harvested sustainably?’. At that time, the answer was more or less reasoned to be an affirmative, although there were still considerations and concerns (mostly focused on the harvest of vulnerable LPS taxa). The international marine aquarium industry and Australian coral harvest fisheries between 2003 and the present (2024) are vastly different in geographic scale, total harvest, and economic value. Instead of a curio and live rock dominated harvest within discreet zones (Hariott, 2001), harvest is now composed of live ornamental corals, dominated by SPS *Acropora* and a selection of ever popular, iconic Australian LPS species (Chapter 2; Pratchett, 2021a). What then, is the answer to Harriott (2003)’s same question if posed today? In short, the answer to this question is unknown, and more elusive and difficult to answer than it was two decades ago. Current harvest quota lacks ecologically relevant justification, and no Australian states have the capacity to quantitatively, accurately, reliably, and irrefutably assess the sustainability of past, current, and potential future coral harvest. Ultimately, whether an answer to Harriott (2003)'s classic question is ever obtained is dependent on the willingness to invest in the necessary research and management 'tailoring'. Though difficult to estimate with available information on current investment and allocation of funds for these fisheries, the amount required to bridge the existing vast knowledge and methodology gaps between current harvest management, and quantitative harvest management, is presumably immense; at least based off the challenges and shortcomings highlighted in current knowledge throughout this work (but esp. Chapter 5). Due to lack of supporting quantitative evidence for non-detriment, the major threat to the economic viability of these industries appears to be a reduction or exclusion of market access as a result of trade restrictions of international origin due to unsubstantiated claims of non-detrimental impact (European Commission, 2023; Gay, 2023, 2022). With further investment into properly adjusting these mechanisms to the unique challenges of managing coral harvest, Australia has the opportunity to justify its historic reputation for sustainable coral harvest fisheries (Chapter 1 and 2; Harriott, 2003, 2001) and crystalise its position as an irrefutable global standard setter for the sustainable management of commercial ornamental aquarium coral harvest.

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Appendices

Appendix A: Chapter 2

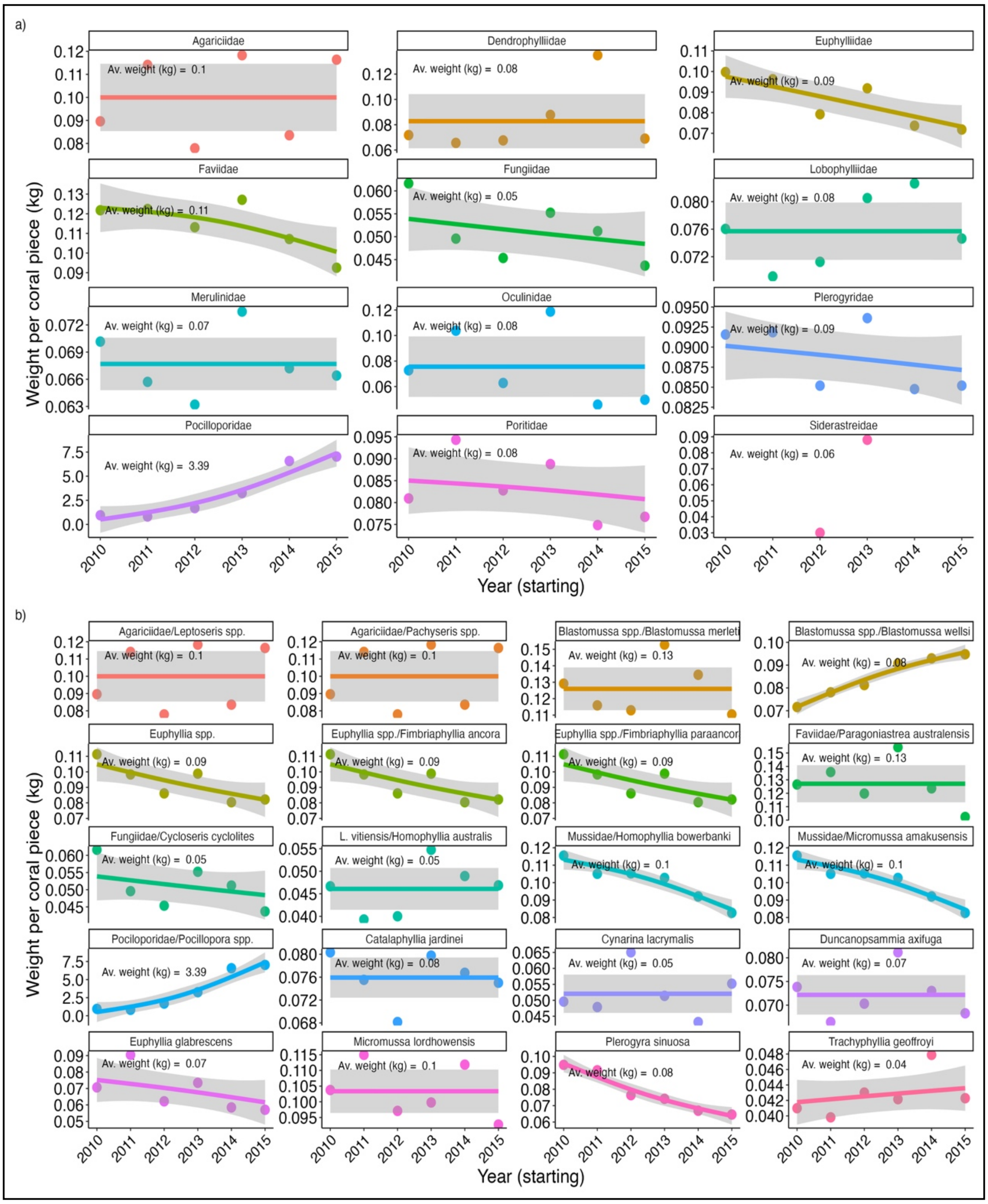

Figure A1. Average weight per piece at the a) family level and b) genus/species level from the financial year starting 2010 (i.e., 2010/11) to the financial year starting 2015 (i.e., 2015/16), with overall average weight per piece value over the defined period for select sclarctinian coral taxa harvested in the QCF (Queensland Coral Fishery).

Appendix B: Chapter 3

Table B1. Additional user defined statistical parameters used to model the size weight relationship of four distinct *Acropora* growth form groups (corymbose/caespitose, digitate, staghorn/bottlebrush, table) including A) table of model prior parameters, and B) model validation results (leave one out/LOO model comparison). In Table A), the column ‘dist’ describes the distribution used on each prior, with ‘prior’ describing the user defined prior values. The prior ‘class’ identifies the prior type, ‘b’ or ‘sigma’, with sigma representing the error term. The ‘coef’ column identifies which prior was the intercept term for the ‘nlpar’ (i.e., non-linear parameter) defined as either the ‘a’ or ‘b’ term of the two-factor power relationship formula. The ‘source’ identifies the origin of values, either being user defined, vectorized from user definitions, or default. In table B),’elp_diff’ represents elapsed difference and ‘st_diff’ as the standard error of difference between models according to leave one out (LOO) model comparison. For further explaination of non-linear Baysian statistical inference, see Bürkner (2017a,b).

Table of model prior parameters.

All models run using seed = 1234, chains = 4, cores = 4, iter = 12000, warmup = 5000, thin = 12.

model	**dist**	**prior**	**class**	**coef**	**nlpar**	**source**
Cor./Caes.						
	normal	0.1, 0.1	b		‘a’	user
	normal	0.1, 0.1	b	Intercept	‘a’	vectorized
	normal	0.5, 0.1	b		‘b’	user
	normal	0.5, 0.1	b	Intercept	‘b’	vectorized
	Student_t	3, 0, 2.5	sigma			default
Digitate						
	normal	0.5, 0.5	b		‘a’	user
	normal	0.5, 0.5	b	Intercept	‘a’	vectorized
	normal	0.7, 0.5	b		‘b’	user
	normal	0.7, 0.5	b	Intercept	‘b’	vectorized
	Student_t	3, 0, 2.5	sigma			default
Stag./Bott.						
	normal	0.5, 0.5	b		‘a’	user
	normal	0.5, 0.5	b	Intercept	‘a’	vectorized
	normal	0.7, 0.5	b		‘b’	user
	normal	0.7, 0.5	b	Intercept	‘b’	vectorized
	Student_t	3, 0, 2.5	sigma			default
Table		0.5, 0.5	b		‘a’	user

	normal					
	normal	0.5, 0.5	b	Intercept	‘a’	vectorized
	normal	0.7, 0.5	b		‘b’	user
	normal	0.7, 0.5	b	Intercept	‘b’	vectorized
	Student_t	3, 0, 2.5	sigma			default

A) Validation results

Lognormal vs normal

	elpd_diff	se_diff
Lognormal	0.0	0.0
Normal	-8161.6	618.5

Power vs exponential vs linear

	elpd_diff	se_diff
Power	0.0	0.0
Linear	-250.0	40.9
Exponential	-587.4	69.3

Appendix C: Chapter 4

Table C1. Table showing estimate and standard error (*SE*) of each non-linear parameter constant, '*a*' (i.e. scaling factor) and '*b*' (i.e. exponent), along with lower (L) and upper (U) 95% Highest Density Intervals (HDIs) for each species. Estimates and *SE* were extracted using the 'as_draws_rvars' function of the 'posterior' package (Bürkner et al., 2021). HDIs were extracted using the 'ci' function of the package 'bayestestR' (Makowski et al., 2019).

Growth Form	**estimate ± *SE***	**L. 95% HDI**	**U. 95% HDI**
Non-linear constant '*a*'			
Catalaphyllia jardinei	0.69 ± 0.55	0.01	1.76
Duncanopsammia axifuga	0.20 ± 0.038	0.13	0.27
Euphyllia glabrescens	0.43 ± 0.11	0.25	0.67
Homophyllia australis	0.73 ± 0.14	0.48	1.01
Micromussa lordhowensis	1.83 ± 0.23	1.42	2.31
Trachyphyllia geoffroyi	0.36 ± 0.033	0.30	0.42
Non-linear constant '*b*'			
Catalaphyllia jardinei	2.73 ± 0.35	2.09	3.41
Duncanopsammia axifuga	2.77 ± 0.06	2.65	2.89
Euphyllia glabrescens	2.74 ± 0.094	2.54	2.90
Homophyllia australis	2.59± 097	2.38	2.77
Micromussa lordhowensis	2.06 ± 0.045	1.97	2.15
Trachyphyllia geoffroyi	2.89 ± 0.037	2.82	2.97

Table C2. Additional size-weight model parameters including seed, chains, cores, iterations (iter), warmup, thin, and table showing LPS coral species and model priors. The column 'dist' describes the distribution used on each prior, with 'prior' describing the user defined prior values. The prior 'class' identifies the prior type, 'b' or 'sigma', with sigma representing the error term. The 'coef' column identifies which prior was the intercept term for the 'nlpar' (i.e., non-linear parameter) defined as either the 'a' or 'b' term of the two-factor power relationship formula. The 'source' identifies the origin of values, either being user defined, vectorized from user definitions, or default.

All models run using seed = 1234, chains = 4, cores = 4, iter = 12000, warmup = 5000, thin = 12, distributional family = gaussian.

model	**dist**	**prior**	**class**	**coef**	**nlpar**	**source**
Catalaphyllia jardinei	normal	0.5, 1.5	b		'a'	user
	normal	0.5, 1.5	b	Intercept	'a'	vectorized
	normal	3, 1	b		'b'	user
	normal	3, 1	b	Intercept	'b'	vectorized
	Student_t	3, 0, 45.2	sigma			default
Duncanopsammia axifuga	normal	0.5, 0.3	b		'a'	user
	normal	0.5, 0.3	b	Intercept	'a'	vectorized
	normal	3, 0.4	b		'b'	user
	normal	3, 0.4	b	Intercept	'b'	vectorized
	Student_t	3, 0, 123.1	sigma			default
Euphyllia glabrescens	normal	1.4, 0.5	b		'a'	user
	normal	1.4, 0.5	b	Intercept	'a'	vectorized
	normal	2.4, 0.6	b		'b'	user
	normal	2.4, 0.6	b	Intercept	'b'	vectorized
	Student_t	3, 0, 99.3	sigma			default
Homophyllia australis	normal	1, 0.5	b		'a'	user
	normal	1, 0.5	b	Intercept	'a'	vectorized
	normal	2.3, 0.4	b		'b'	user
	normal	2.3, 0.4	b	Intercept	'b'	vectorized
	Student_t	3, 0, 34.1	sigma			default
Micromussa lordhowensis	normal	1, 0.8	b		'a'	user
	normal	1, 0.8	b	Intercept	'a'	vectorized
	normal	2.5, 0.5	b		'b'	user
	normal	2.5, 0.5	b	Intercept	'b'	vectorized

	Student_t	3, 0, 207.6	sigma			default
Trachyphyllia geoffroyi	normal	0.5, 0.1	b		‘a’	user
	normal	0.5, 0.1	b	Intercept	‘a’	vectorized
	normal	3, 0.1	b		‘b’	user
	normal	3, 0.1	b	Intercept	‘b’	vectorized
	Student_t	3, 0, 112.7	sigma			default
Species Comparison (“ANCOVA”) model	normal	1.5, 0.8	b			
	normal	1.5, 0.8	b	Intercept	‘a’	user
	normal	0, 1.6	b		‘a’	vectorized
	normal	0.45, 0.04	b	Intercept	‘b’	user
	Student_t	3, 0, 2.5	sigma		‘b’	vectorized
	normal	0, 1.6	b	DMAX.cm	‘c’	vectorized
	normal	0, 1.6	b	DMAX.cm:speciesDuncanopsammiaaxifuga	‘c’	vectorized
	normal	0, 1.6	b	DMAX.cm:speciesEuphylliaglabrescens	‘c’	vectorized
	normal	0, 1.6	b	DMAX.cm:speciesHomophylliaaustralis	‘c’	vectorized
	normal	0, 1.6	b	DMAX.cm:speciesMicromussalordhowensis	‘c’	vectorized
	normal	0, 1.6	b	DMAX.cm:speciesTrachyphylliageoffroyi	‘c’	vectorized
	normal	0, 1.6	b	Intercept	‘c’	vectorized
	normal	0, 1.6	b	speciesDuncanopsammiaaxifuga	‘c’	vectorized
	normal	0, 1.6	b	speciesEuphylliaglabrescens	‘c’	vectorized
	normal	0, 1.6	b	speciesHomophylliaaustralis	‘c’	vectorized
	normal	0, 1.6	b	speciesMicromussalordhowensis	‘c’	vectorized

	normal	0, 1.6	b	speciesTrachyphyllia geoffroyi	‘c’	user
	normal	0.45, 0.04	b		‘c’	vectorized

Appendix D: Chapter 5

Table D1. Additional logistic regression parameters including seed, chains, cores, iterations (iter), warmup, thin, and table showing logistic regression model priors (where dist = distribuition, coef = coefficient). The column ‘dist’ describes the distribution used on each prior, with ‘prior’ describing the user defined prior values. The prior ‘class’ identifies the prior type, ‘b’ or ‘sigma’, with sigma representing the error term. The ‘coef’ column identifies which prior was the intercept term for the ‘nlpar’ (i.e., non-linear parameter) defined as either the ‘a’ or ‘b’ term of the two-factor power relationship formula. The ‘source’ identifies the origin of values, either being user defined, vectorized from user definitions, or default.

All models run using seed = 1234, chains = 4, cores = 4, iter = 18000, warmup = 4000, thin = 18, distributional family = binomial(logit link).

model	dist	prior	class	coef	source
GBR					
	normal	0, 200,000	b		user
	normal	0, 200,000	b	scaleaverage.biomass.pm2	vectorized
	normal	0, 200,000	b	scalefishing_mortality	vectorized
	normal	0, 200,000	b	scalemax.biomass.pm2	vectorized
	normal	0, 200,000	b	scaletotal.reef.area.m2	vectorized
	normal	0, 1500	Intercept		user
Cairns					
	normal	0, 200,000	b		user
	normal	0, 200,000	b	scalefishing_mortality	vectorized
	normal	0, 1500	Intercept		user