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**THE ECOLOGY OF SHARK-LIKE BATOIDS: IMPLICATIONS FOR
MANAGEMENT IN THE GREAT BARRIER REEF REGION**

Thesis by
Jimmy White B.Sc. (Hons)
Submitted

For the degree of Doctor of Philosophy
in the School of Earth and Environmental Sciences
James Cook University
Townsville

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STATEMENT OF CONTRIBUTION

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CA, Tobin, AJ. Oceania Chondrichythan Society, Gold Coast 2011

Assessing the susceptibility of *Glaucostegus typus*, *Rhynchobatus australiae* and
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CA, Tobin, AJ. Australian Society of Fish Biology, Townsville 2011

Assessing the susceptibility of *Glaucostegus typus*, *Rhynchobatus australiae* and
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ABSTRACT

Shark-like batoids are a group of elasmobranchs with a body form similar to that of sharks (i.e. elongate body, well developed caudal and dorsal fins), but with head, gill and mouth morphology similar to that of skates and stingrays. Severe population declines of and reduction in geographic distribution throughout the South-East Asian portion of their range suggest *Glaucostegus typus* and *Rhynchobatus* spp. are vulnerable to depletion by unregulated take in fisheries. Shark-like batoids in fisheries have been poorly studied in comparison to sharks and data on these populations is severely lacking. Given the value of their fins, fishing is likely to continue in many locations. This research provides knowledge of their life history, ecology and how they interact with fisheries. This information is essential for informing decision making tools and the development of successful management strategies.

Between 2007 and 2009, an on board vessel observer program was conducted to examine the composition of elasmobranch catch in Queensland's east coast inshore finfish fishery (ECIFF). Of the batoids, *G. typus* were most frequently caught in intertidal habitats, whereas *Rhynchobatus* spp. dominated the catch in inshore coastal habitats. Comparison of gill-net catches to research long-line sampling showed that not all size classes of shark-like batoids are captured by the gill-net fishery. Current mesh size restrictions in Queensland's gill-net fisheries limited interaction with *G. typus* to juveniles. Given that home-range size and habitat use by elasmobranchs can vary between ontogenetic stages and species, vulnerability to fisheries may vary depending on overlap of preferred habitats and fishing activity, and whether each size class is susceptible to the gear.

Given the naturally low abundances of shark-like batoids, ecological sensitivity and structural complexity of some habitats within the Great Barrier Reef Marine Park (GBRMP), the use of conventional fisheries independent sampling across such a large geographic area would be impossible. The applicability of baited remote under-water video station (BRUVS[®]) to determine the distribution of elasmobranchs at broad spatial scales (12° latitude) and across a range of habitat types was assessed.

This broad assessment allowed examination of distribution at the ecosystem scale in environments outside of those typically targeted by commercial gill-net fisheries. Shark-like batoids were observed across a wide depth range (10.4 - 87.5 m), showing highest affinity for 30 - 40 m. The depth range of highest affinity is outside the maximum depth at which the ECIFF operates. It is likely that preference for these deeper habitats may limit

exposure of *Rhynchobatus* spp. to the commercial gill-net fishery. There was no difference in the proportion of BRUVS[®] in which shark-like batoids were observed between fished and unfished marine park zones. BRUVS[®] results suggest that marine protected areas (MPAs) may be of limited benefit to mobile habitat generalists like *Rhynchobatus* spp. Baited remote under-water video station surveys offer a standardized, non-extractive technique for quantifying the spatial distribution of mobile species that are difficult to sample using conventional techniques across broad spatial scales.

The utility of BRUVS[®] was severely limited in high turbidity environments, such as inshore coastal embayment's within the GBRMP. Acoustic telemetry was used in these habitats to examine the space use and residency of shark-like batoids. *Glaucostegus typus* were monitored between 1 and 766 days (mean = 333 ± 69 days) and were present in the site from 1 to 198 days (mean 73 ± 25 days). Both adult male and female *G. typus* exhibited philopatric behaviour patterns, leaving the bay and returning after periods of about 9 -12 months to use the same areas where they were detected in previous years. Strong site fidelity observed in adult *G. typus* suggests that nearshore areas are a key component of the species' spatial ecology, and may form critical habitat. Strong philopatry evident in adult *G. typus* suggest this species may benefit from the use of discrete areas of protection in inshore coastal waters during austral summer months. *Rhynchobatus* spp. were monitored for 1 to 707 days (mean = 231 ± 50 days) and were present in the site from 1 to 350 days (mean 82 ± 24 days). *Rhynchobatus* spp. exhibited no synchronicity in use of the bay and size had no effect on residency. These findings support those from the BRUVS[®], *Rhynchobatus* spp. is likely a habitat generalist with any patterns of spatial ecology possibly diluted by the presence of three species.

The life histories of *G. typus* and *Rhynchobatus* spp. captured in the ECIFF were examined using vertebral ageing. The sigmoid growth functions, Gompertz and logistic, best described the growth of *Rhynchobatus* spp. and *G. typus*, providing the best statistical fit and most biologically appropriate parameters. The 2-parameter logistic was the preferred model for *Rhynchobatus* spp. with growth parameter estimates (both sexes combined); $L_{\infty} = 2045$ mm STL; $k = 0.41 \text{ yr}^{-1}$. The 2-parameter logistic growth model was also the preferred model for *G. typus* with growth parameter estimates (both sexes combined); $L_{\infty} = 2770$ mm STL; $k = 0.30 \text{ yr}^{-1}$. Annual growth-band deposition could not be excluded in *Rhynchobatus* spp. using mark-recaptured individuals. A single growth curve has been proposed for the *Rhynchobatus* spp. complex, given biological samples have been pooled between complex members. Further the complex is currently managed as a single species and a single growth model may prove useful in informing future management strategies. Although morphologically similar *G. typus* and *Rhynchobatus*

spp. have differing life histories, with *G. typus* longer lived, slower growing and attaining a larger maximum size.

Sensitivities to uncertainties in biological parameters and vulnerability to fisheries depletion of *G. typus* and *Rhynchobatus* spp. were examined using population matrix models. Unfished *G. typus* and *Rhynchobatus* spp. had robust populations with positive population growth (λ), of 1.38 yr^{-1} and 1.27 yr^{-1} and generation times between 7.05 and 7.54 years, respectively. Increasing longevity of *G. typus* had little effect on rates of population increase, while increasing natural mortality by 50 % reduced rates of population increase λ , from 1.55 yr^{-1} to 1.22 yr^{-1} . Increased age at maturity produced the lowest rates of population increase λ , 1.13 yr^{-1} and longest generation times (12.0 years). *Rhynchobatus* spp. were robust to increases in longevity; however increasing natural mortality by 50 % rapidly decreased rates of population growth λ , from 1.28 yr^{-1} to 1.01 yr^{-1} . Models with earlier ages at maturity had higher rates of population growth $\lambda = 1.27 \text{ yr}^{-1}$ in comparison to models with older ages at maturity ($\lambda = 1.04 \text{ yr}^{-1}$). Population models were also used to examine the effectiveness of alternative management strategies to the current initiatives in place in Queensland waters. Minimum size limits performed best for *G. typus*, while maximum size limits were the most beneficial management strategy for *Rhynchobatus* spp. The difference in strategies was the result of variation in life history between the species. Based on the results of demographic analysis, size limits may be a suitable management tool for shark-like batoids assuming they could be effectively implemented and may offer greater protection than current bag limit strategies that are in place for *Rhynchobatus* spp.

The findings from this dissertation highlight the need for species specific investigations of biology, spatial ecology and fisheries interaction. Despite morphological similarities between focal species their habitat use and biology are different. This dissertation has provided substantial information necessary in both the assessment of shark-like batoid vulnerability within the GBRMP in addition to the development of targeted species-specific management strategies. Managing for individual species in multi-species fisheries is challenging. Understanding how and where fisheries interactions occur and the vulnerability of populations to these interactions is essential if bycatch species are to receive effective targeted management in mixed species fisheries.

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CHAPTER 1

General Introduction



Plate 1. Remembering to enjoy the little things. Photo by Fernanda DeFaria (Cleveland Bay, 2010).

Despite a poor understanding of the biology and ecology of shark-like batoid species they have been heavily exploited by both artisanal and commercial fisheries as either a target or byproduct species (White and McAuley 2003a). The meat is sold for human consumption but it is the fins that are prized, known as “white-fin”, it is among the most lucrative of elasmobranch products (White and McAuley 2003a). The high value fins, particularly from large animals, give fishers strong incentive to retain captured individuals (White and McAuley 2003a). Shark-like batoid are susceptible to multiple gear types including trawl nets, gill-nets, trap and seine nets and hooks (Bentley 1996a, Chen 1996, White and McAuley 2003a, b). Fishing effort is particularly intense in South-East Asia (Bentley 1996a, Chen 1996) where reductions in population size have been inferred from declines in catch rates reported by the gill-net fishery fleet (White and McAuley 2003a). With evidence of population decline, continued high levels of exploitation particularly in South-East Asia, and growing demand for fins, both *G. typus* and *R. australiae* have been assessed as Vulnerable A2bd+3bd+4bd by the International Union for Conservation of Nature (IUCN) (White and McAuley 2003a, b).

In Australian waters, batoids are not targeted, however they are known bycatch of trawl (Stobutzki et al. 2002, Stephenson and Chidlow 2003) and gill-net fisheries (Halliday et al. 2001). Depletion of batoid populations in Australian fisheries may therefore go unnoticed as target species with higher production rates continue to support fisheries (Musick 1999, Stevens et al. 2000a). Batoids as a group represent 1.75% of the total catch composition for Queensland’s inshore net fishery (Harry et al. 2011b). *Glaucostegus typus* and *Rhynchobatus* spp. account for 15.4% and 39.8% of the fisheries batoid catch and are amongst the top twenty most frequently caught elasmobranch species (Harry et al. 2011b). Presently both *G. typus* and *R. australiae* are assessed by the IUCN as Near Threatened in Australian waters. Given population declines (particularly of *R. australiae*) throughout South-East Asia and the high value placed on fins this classification may be upgraded. In Australia, management strategies initiated to mitigate anthropogenic impacts on batoids (fishing activity and habitat degradation) have to a large extent been precautionary. Strategies include catch limits on *Rhynchobatus* spp., changes to finning laws for all elasmobranch species and networks of marine protected areas.

As elasmobranchs, batoids are thought to have a K-selected life history strategy which includes slow growth, late maturity and production of small numbers of large young (Walker 1998, Stevens et al. 2000a). Species with K-selected life histories have limited resilience to fishing mortality (Hoenig and Gruber 1990, Musick and Bonfil 2004). There is little information regarding batoid life history strategies, which limits the current understanding of how these populations will respond to fishing pressure. Much of what is

currently known is based on data from sharks. Within shark fisheries, there is considerable variability in life history strategies of targeted species (Cortes 2000), and therefore variability in vulnerability to fishing pressure. Examples of elasmobranch fisheries collapse have generally come from large, slow growing, temperate species (Simpfendorfer 1999a). However, not all sharks fall into this category, for instance some tropical sharks, have life histories that may be more resilient to exploitation (small size, short-lived, fast maturing) (Simpfendorfer 1999). By quantifying the life history characteristics of batoids, a greater understanding of their vulnerability to fishing pressure can be defined to help develop appropriate conservation and management strategies.

Demographic population analysis is a useful tool for effectively estimating and managing elasmobranch stocks (Chen and Yuan 2006). By determining a population's rate of change (growth/decline) it is possible to characterize its vulnerability to fishing (Frisk et al. 2005). There are two forms of demographic models commonly applied to elasmobranch populations, static population assessments and rebound potentials. Static assessments use either life-tables or Leslie matrices and can be conducted based solely on the life history traits of the species (Cortes 2004). Rebound potentials take this a step further by modifying life tables to incorporate density dependence. The latter method allows the capacity of species to recover from fishing pressure to be quantified (Au and Smith 1997, Smith et al. 1998).

The spatial ecology of batoids within nearshore tropical environments remains a poorly studied topic. Nearshore systems are highly dynamic environments that experience large variations in abiotic parameters (temperature, salinity, turbidity) as well as anthropogenic influences (pollution, habitat degradation and fishing activity) affecting both the habitat and its inhabitants. Nearshore environments are also key centres of activity for intensive recreational and commercial fishing activities due to their proximity to coastal communities and high productivity. Additionally, intense coastal development such as dredging, construction, erosion and deforestation, contribute to large-scale habitat alteration or destruction (Edgar et al. 2001). Nearshore environments are where batoid species are most likely to be adversely affected by anthropogenic impacts. Understanding where these species move within nearshore systems, how long they are present and why they are present are necessary first steps toward assessing the effectiveness of current management strategies, such as fishing closures and marine protected areas and the development of future strategies designed to mitigate the impact of anthropogenic influences.

Marine protected areas as a fisheries management tool have been applied to a variety of resources, most commonly teleosts from tropical and temperate waters. However, there is limited experience in the usage of MPAs for the protection or enhancement of elasmobranch species (Kinney and Simpfendorfer 2009). While some areas within the Great Barrier Reef Marine Park (GBRMP) have been protected from anthropogenic disturbance (including fishing) for over 25 years, the majority of this protected area was limited to coral reef habitat. The Great Barrier Reef Marine Park Authority (GBRMPA) recently re-zoned the park based on closed areas designated by habitat type or bioregions (Fernades et al. 2005). This re-zoning resulted in large tracts of representative habitats (bioregions) within the GBRMP being closed to fishing and designated as protected zones. Protection zones were determined based on a precautionary approach and were made with imperfect knowledge of populations within these habitats (Fernades et al. 2005).

To be effective, MPAs must include a mix of suitable habitats that provide protection to all life stages of the species in question (Bonfil 1999). It is possible that no-take MPAs can be an effective way to support and complement the conservation of some shark and batoid species, providing a haven during key parts of their life cycles. However, for large mobile batoid species with poorly defined patterns of movement, limited data on habitat use and potential life stage partitioning, defining the area needed to provide protection is difficult. If individuals are continually moving out of an MPA and exposed to fishing, the efficacy of the region as refuge from the surrounding fishery falls into question. To understand the utility of MPAs for these species requires basic biological and ecological data defining the size of home ranges, habitat utilisation patterns, migratory routes, timing of movements, and the amount of movement in and out of existing MPAs. Without this information, it is not possible to quantify the extent to which existing zoning of the GBRMP is protecting nearshore habitats and its inhabitants.

Passive acoustic monitoring is a powerful tool capable of providing long-term data that elucidates subtle changes in behaviour and habitat use without the logistical constraints and potential bias associated with active tracking. Commonly researchers using this technology design single species studies and use acoustic receivers to “acoustically recapture” individuals, thus generating data similar to traditional animal telemetry or mark-recapture studies (White and Garrott 1990). The present project will use this approach to monitor batoid species to examine their interactions with nearshore habitats. Previous studies have shown that this type of research can be used to address complex questions related to marine policy analysis and the efficacy of marine zoning (Heupel and Simpfendorfer 2005), responses to environmental change and responses to natural and

anthropogenic disturbance (Heupel et al. 2003, Heupel et al. 2008). This project will address current knowledge gaps and examine the efficacy of MPAs as a management tool for highly mobile, nearshore batoids by monitoring their presence, movement patterns and inter-specific interactions.

There is growing concern for the sustainability of elasmobranchs in fisheries globally (Hoenig and Gruber 1990, Stevens et al. 2000a, Graham et al. 2001, Clarke et al. 2006b, Dulvy et al. 2008). Low biological productivity of many species (Pratt and Casey 1990, Walker 1998, Stevens et al. 2000a), combined with oftentimes high catch susceptibility, means that many elasmobranch species require management action long before sufficient data are available to undertake full stock assessment (Walker 2004). The situation for species taken as bycatch may be worse, particularly for those less able to sustain their populations under fishing regimes designed to sustain stocks with r-selected life history strategies (e.g. teleost and invertebrates) (Hueter 1998). If elasmobranchs are to be effectively managed as target or bycatch species, knowledge of when and where they are available to fisheries are essential for developing sustainable harvest strategies (McAuley et al. 2007a). Knowledge of seasonality can provide the basis for seasonal closures (Walker 1999, Hunter et al. 2006b), while knowledge of what habitats they are more likely to inhabit can facilitate gear restriction or spatial closure management (Williams and Schaap 1992, Walker and Hislop 1998b, Simpfendorfer 1999b), should mitigative management intervention be required.

Globally, batoids have become an increasingly large component of fisheries catch, and in some cases have developed into target species in fisheries where they were once considered by-product or bycatch (Anon 2003, White and Dharmadi 2007). Skates have traditionally been landed for their flesh (Holden 1973, 1974, Walker and Heessen 1996, Walker and Hislop 1998a), whereas shark-like batoids (families Rhinobatidae, Rhynchobatidae, Rhinidae, Pristidae) are taken for both flesh and fins; especially in South-East Asia. Fins of shark-like batoids, known as “white-fin”, are highly prized and among the most lucrative of elasmobranch products (White and McAuley 2003a, Clarke et al. 2006a, Clarke et al. 2006b, Compagno et al. 2006b). Shark-like batoids in fisheries have been poorly studied in comparison to sharks and data on these populations is severely lacking. Given the value of their fins, fishing is likely to continue in many locations. Thus research that helps to improve knowledge of their life history, ecology and how they interact with fisheries and fishing gear will be valuable in improving management of this group of species.

Their shark-like bodies are unlike the disc shaped body plans of most batoids (e.g. Zanobatidae, Platyrrhinidae, Myliobatidae) and may increase their susceptibility to capture in some fisheries gear. These species are caught by a variety of fishing gears including trawl, gill-net, trap and seine nets and hooks (Bentley 1996a, Chen 1996, White and McAuley 2003a, Compagno et al. 2006b). Fishing effort for *G. typus* and *Rhynchobatus* spp. is particularly intense in South-East Asia (Bentley 1996a, Chen 1996) where reductions in population size have been inferred by declining catch rates in gill-net fisheries (White and McAuley 2003a, b). There are no target fisheries for shark-like batoids in Australian waters; however *G. typus* and *Rhynchobatus* spp. are caught in trawl (Stobutzki et al. 2002), seine and gill-net fisheries (Harry et al. 2011b). *Glaucostegus typus* has been assessed as Vulnerable by the International Union for Conservation of Nature (IUCN) (White and McAuley 2003a). This assessment is the result of intense fishing pressure, particularly in Indonesian waters, and a fall in catch records from fisheries operating in these regions (White and McAuley 2003a). In Australia *G. typus* has been classified as 'high risk' due to distributional overlap with multiple fisheries (notably gill-net and penaid trawl) and low productivity (Salini et al. 2007).

A complex of Rhynchobatidae have been reported in Australian waters consisting of three distinct species (*Rhynchobatus australiae*, *R. laevis* and *R. palpebratus*), which until now have been consistently confused in the literature (Last and Stevens 2009). Although each of the species within the complex has been assessed as Vulnerable by the International Union for Conservation of Nature (IUCN) the extent of individual species decline and range reductions are hard to quantify given taxonomic confusion (McAuley and Compagno 2003b, White and McAuley 2003a, Compagno and Marshall 2006). Current management strategies within Queensland waters treat the species complex as a single group despite preliminary evidence of varying size and ages at maturity and maximum sizes (Last and Stevens 2009). Moreover, considering identification difficulties will persist management is likely to continue treating these species as a complex. Hence, the present study has treated all individuals as a group and will herein be referred to as *Rhynchobatus* spp.

1.1 Project aims and objectives

This project will investigate the biology, spatial ecology and fisheries interactions of *G. typus* and *Rhynchobatus* spp. in a nearshore marine environment. Collected biological data will define demographic parameters of species through the determination of life history (age structure, growth) parameters, population structure and life stage partitioning across temporal and spatial scales. Spatial analysis based on acoustic monitoring will

identify differential use of available habitats within and among species, life history stages and seasons. These data will be used to investigate the amount of protection MPAs designated by the Great Barrier Reef Marine Park Authority provide for these high-value batoid species.

Thesis objective:

Define the fisheries interaction, biology and spatial ecology of *G. typus* and *Rhynchobatus* spp. in nearshore waters.

Specific objectives:

1. Determine the incidence and availability of *G. typus* and *Rhynchobatus* spp. to commercial gill-net fisheries within the Great Barrier Reef
2. Investigate the amount of refuge from fishing pressure provided to large, mobile batoid species by marine protected areas
3. Determine life history parameters of age and growth for *G. typus* and *Rhynchobatus* spp.
4. Examine population demography and effectiveness of management scenarios for *G. typus* and *Rhynchobatus* spp.

1.2 Source of data

Biological samples and fisheries dependent catch data for this research were obtained from the on board vessel observer survey program undertaken as part of the Marine and Tropical Sciences Research Facility (MTSRF) Project 4.8.4, in which the author was one of three main observers collecting data. Biological samples in addition to animals for acoustic monitoring were obtained via fisheries independent sampling. Vertebral samples of *G. typus* were provided by Dr. Will White. The BRUVS[®] data used in chapter 3 is an output from the 'Great Barrier Reef Seabed Biodiversity Project'; a collaboration between the Australian Institute of Marine Science (AIMS), the Commonwealth Scientific and Industrial Research Organization (CSIRO), Queensland Primary Industries & Fisheries (Department of Employment, Economic Development and Innovation (DEEDI), formerly QDPIF) and the Queensland Museum (QM). The project was funded by the CRC Reef Research Centre, the Fisheries Research and Development Corporation (FRDC) and the National Oceans Office, and led by Drs R. Pitcher (Principal Investigator, CSIRO), P. Doherty (AIMS), J. Hooper (QM) and N. Gribble (QDPIF).

1.3 Thesis outline

Chapter 2 Examines the incidence and availability of shark-like batoids to a commercial gill-net fishery operating within the Great Barrier Reef World Heritage Area. Comparisons between fisheries dependent and independent data allows determination of catch probabilities for species between habitats in which the fishery operates and seasons.

Chapter 3 Examines the utility of Baited Remote Underwater Video Surveys (BRUVS[®]) to investigate the geographic distribution and habitat preference of shark-like batoids within the Great Barrier Reef Marine Park (GBRMP). Incidence of shark-like batoids between marine park management zones and variations of habitat preference with season were also examined.

Chapter 4 Investigates the fine scale habitat preferences and residency of shark-like batoids within a nearshore system. Using data from long-term acoustic monitoring, home range size and indices of residency were calculated to assess interspecific differences in spatial ecology in addition to intraspecific differences associated with sex and size.

Chapter 5 Provides preliminary estimates of age and growth for the shark-like batoid species, *G. typus* and *Rhynchobatus* spp. based on biological samples obtained from Queensland's commercial gill-net fishery.

Chapter 6 Uses demographic population analysis to examine sensitivity to uncertainty in life history parameters, vulnerability of shark-like batoids to fishing mortality and evaluate alternative management strategies.

CHAPTER 2

Incidence and availability of shark-like batoids of conservation concern to fishing



Plate 2. Observing gill-net catch from inshore fishing grounds. Photo taken by Sterling Peverell (Old Mapoon, 2009).

2.1 Introduction

There is growing concern for the sustainability of elasmobranchs in fisheries globally (Hoenig and Gruber 1990, Stevens et al. 2000a, Graham et al. 2001, Clarke et al. 2006b, Dulvy et al. 2008). Low biological productivity (Pratt and Casey 1990, Walker 1998, Stevens et al. 2000a), combined with high catch susceptibility means that many elasmobranch species require management action before sufficient data are available to undertake a full stock assessment (Walker 2004). The situation for species taken as bycatch may be worse, particularly for those less able to sustain their populations under fishing regimes designed to catch species who can sustain higher levels of removals (e.g. teleost and invertebrates) (Hueter 1998). If elasmobranchs are to be effectively managed as target or bycatch species, knowledge of when and where they are available to fisheries is essential to developing sustainable harvest strategies (McAuley et al. 2007a).

Globally batoids have become an increasingly large component of fisheries catch and in some cases have developed into target species in fisheries where they were once considered by-product or bycatch (Anon 2003, White and Dharmadi 2007). Skates have traditionally been landed for their flesh (Holden 1973, 1974, Walker and Heessen 1996, Walker and Hislop 1998a), whereas shark-like batoids (families Rhinobatidae, Rhynchobatidae, Rhinidae, Pristidae) are taken for both flesh and fins. Fins of shark-like batoids, known as “white-fin”, are highly prized and among the most lucrative of elasmobranch products (White and McAuley 2003a, Clarke et al. 2006a, Clarke et al. 2006b, Compagno et al. 2006b). Shark-like batoids are caught by a variety of fishing gears including trawl, gill-net, trap and seine nets and hooks (Bentley 1996b, Chen 1996, White and McAuley 2003a, Compagno et al. 2006b), but their presence in fisheries has been poorly studied in comparison to sharks, and data on these populations is severely lacking. Fishing effort for shark-like batoids is particularly intense in South-East Asia (Bentley 1996b, Chen 1996) where reductions in population size of *Glaucostegus typus* and *Rhynchobatus* spp. have been inferred by declining catch rates (White and McAuley 2003a, b). There are no target fisheries for shark-like batoids in Australian waters; however rhinobatids, rhynchobatids, rhinids and pristids are caught in trawl and gill-net fisheries (Stobutzki et al. 2002, Zhou and Griffiths 2008, Harry et al. 2011b). Given the value of their fins, fishing of these species is likely to continue in many locations. Thus, research that helps improve knowledge of their life history, ecology, and how they interact with fisheries and fishing gear is valuable in improving management of these species.

Shark-like batoid species taken in Queensland fisheries are considered threatened based on International Union for Conservation of Nature (IUCN) assessments. *Glaucostegus*

typus has been assessed as Vulnerable based on intensive fishing pressure and population declines in South-East Asia (White and McAuley 2003a). Although fishing effort in Australia is not as intense as South-East Asia, *G. typus* has been classified as 'high risk' due to distributional overlap with multiple fisheries (notably gill-net and prawn trawl) and low productivity (Salini et al. 2007). A complex of consistently confounded species (Last and Stevens 2009), from the family Rhynchobatidae (*Rhynchobatus australiae*, *R. laevis* and *R. palpebratus*) have been reported in Australian waters. Although each of the species within the complex has been assessed as Vulnerable by the IUCN, the extent of individual species decline and range reductions are hard to quantify given taxonomic confusion (McAuley and Compagno 2003b, White and McAuley 2003a, Compagno and Marshall 2006). Current management strategies within Queensland waters treat the species complex as a single group despite preliminary evidence of varying size and age at maturity and maximum size (Last and Stevens 2009). Since identification difficulties will persist management is likely to continue treating these species as a complex. Hence, the present study has treated all individuals as a group and will herein be referred to as *Rhynchobatus* spp. Similar to other sawfish (Simpfendorfer 2000, Cavanagh et al. 2003, Carlson et al. 2007), *Anoxypristis cuspidata* has suffered substantial reductions in abundance with populations now fragmented throughout its range (Compagno et al. 2006b). In some regions such as Thailand, Malaysia and Indonesia, *A. cuspidata* has virtually disappeared from commercial catches where it was once encountered regularly (Compagno et al. 2006b). Consequently, *A. cuspidata* is listed as Critically Endangered globally (Compagno et al. 2006b). *Anoxypristis cuspidata* has previously been classified as being at high risk of depletion by fisheries operating in northern Australian waters as they are susceptible to a range of gears, and there is little information regarding their biology (Anon 2003, Peverell 2005, Salini et al. 2007).

The global status of shark-like batoids and the continuing demand for their fins mean that investigation of their interaction with fisheries is important. In this study I examined the incidence of this group of species in a gill-net fishery within the Great Barrier Reef World Heritage Area (GBRWHA), Australia. The aims of the study were to: 1) determine the species and size composition of the shark-like batoid catch in this fishery; 2) examine the influence of habitat and season on capture of these species; and 3) determine the availability of these species to net fishing by comparing the catch results to those from a size-independent fishing gear (long-line).

2.2 Materials and Methods

Fishery dependent sampling

Between March 2007 and December 2009 fishery observers were placed on board vessels operating in the commercial gill-net sector of the East Coast Inshore Finfish Fishery (ECIFF) within the GBRWHA. During surveys observers recorded data on target species, gear type, location, effort and catch composition for each individual net shot. A total of 523 gill-net shots were observed. Shark-like batoids caught were identified, sexed and measured (stretch total length STL) to the nearest cm.

Data were grouped spatially into two nominal habitats (intertidal and inshore coastal) that correspond to discreet sub-components of the ECIFF, each with different target species, fishing practices and management strategies (see Table 2.1). Within intertidal habitats (<2m depth), an array of teleosts are targeted using gill-net of stretched mesh sizes 114-216 mm and lengths up to 600 m. Fishing in intertidal habitats occurs throughout the day/night and throughout the year. In inshore coastal habitats (>2 m depth), teleosts (mostly scombrids) are targeted during winter and spring, whereas sharks are targeted year-round. Up to 600 m of 165 mm stretched mesh gill-net is used.

Table 2.1 Nominal fishery zones (intertidal, inshore coastal) in the East Coast Inshore Finfish Fishery. Table adapted from Harry et al. (2011b).

	Intertidal	Inshore coastal
Depth (m)	0 - 2	2 - 25
Number of nets permitted	3	1
Total gill-net length (m)	600	600
Gill-net mesh size (mm)	114 - 216	165
Gill-net depth (number meshes)	25 - 35	100
Principal target species	<i>Eleutheronema tetradactylum</i> <i>Polydactylus macrochir</i> Mugilidae spp.	<i>Scomberomorus semifasciatus</i> shark

Fishery independent sampling

Fishery independent sampling using long-line gear was conducted to examine shark-like batoid population availability to gill-nets. These data were used to determine whether the

size of shark-like batoids caught in the ECIFF was a function of gear selectivity, or distribution and seasonality of species. Fishery independent sampling was conducted in Cleveland Bay (19°12'3"S, 146°54'4"E); in the central region of the GBRWHA. Sampling was conducted across all seasons and both habitat types between 30th January 2008 and 10th September 2009. Long-lines were 800 m in length consisting of 6 mm rope secured to the bottom by anchors on either end. Hooks were attached to the main line on gangions composed of a 1m section of nylon cord, a swivel and 1 m of wire trace. Circle hooks of two sizes, 14/0 and 16/0 were used. Maximum soak time was 120 minutes. A total of 268 long-line shots were completed. All catch was identified, sexed and measured (STL) to the nearest mm.

Data Analysis

To compare the probability of capture between seasons and habitats with each fishing gear, a full factorial logistic generalised linear model (GLM) with a binomial error structure and logit link function was used. The models included either season (summer: Dec - Feb, autumn: March - Apr, winter: June - Aug, spring: Sept - Nov) or habitat (intertidal, inshore coastal) as factors with an interaction term combining the two. Logistic models were used as they are capable of dealing with inflated zero data typical of catch data. Capture abundance was not incorporated into the models because shark-like batoids were caught in small numbers in comparison to target species within the fishery. Chi squared (χ^2) contingency table were used to compare proportions of shark-like batoids caught between habitats. Two-sample Kolmogorov-Smitnov (KS) tests were used to determine whether length-frequency distributions were significantly different between habitat types and gears.

2.3 Results

Fishery dependent and independent catch composition

One hundred and twenty three shark-like batoids were caught in the 523 observed gill-net shots. *Anoxypristis cuspidata* was the most frequently caught shark-like batoid (44.7 %, n = 55), followed by *Rhynchobatus* spp. (39.8 %, n = 49) and *Glaucostegus typus* (15.4 %, n = 19). Catch composition between habitat types was heterogeneous ($\chi^2 = 40.86$, df = 2, p < 0.001, Fig. 2.1). Both *A. cuspidata* and *G. typus* were more abundant in intertidal than inshore coastal habitats, the reverse was true for *Rhynchobatus* spp. (Fig. 2.1a).

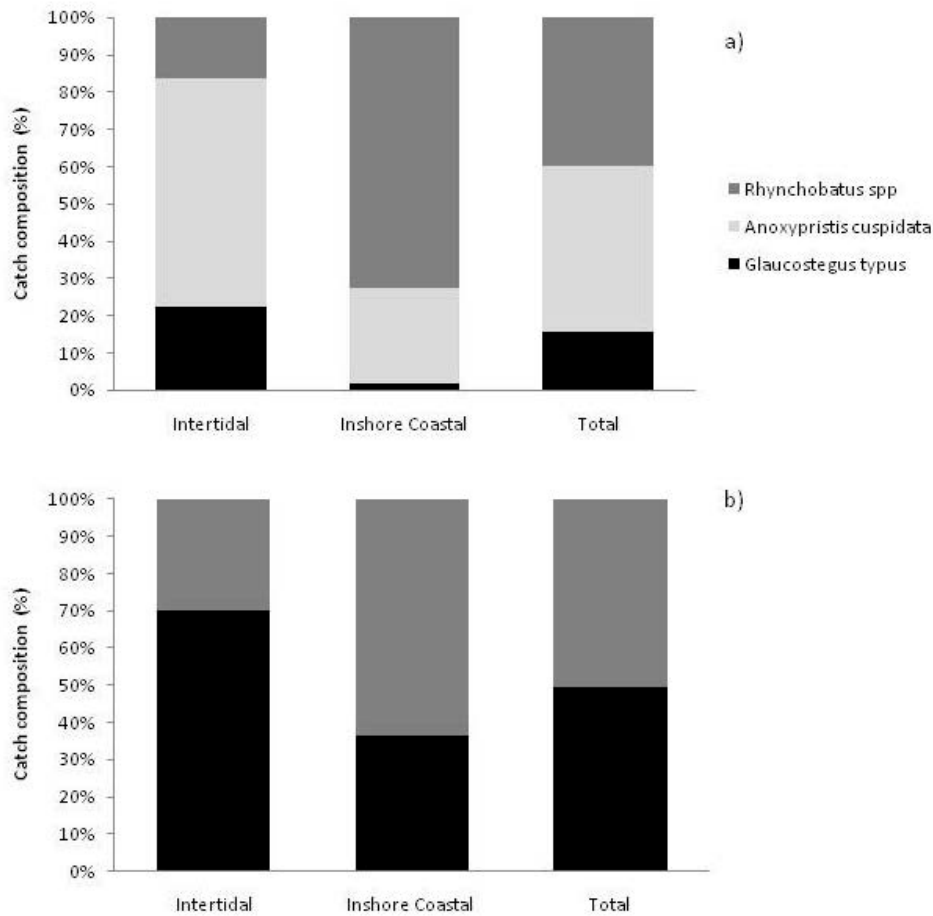


Fig. 2.1 Catch composition of shark-like batoid species from: a) observed commercial gill-net fishery activity within Queensland's East Coast Inshore Finfish Fishery (ECIFF); and b) fisheries independent long-line sampling.

The 261 long-line shots in Cleveland Bay captured 51 *Glaucostegus typus* and 52 *Rhynchobatus* spp. No *Anoxypristis cuspidata* were captured despite many animals ($n = 36$) captures in Cleveland Bay by commercial gill-net activity. Shark-like batoid catch composition was significantly different between habitats ($\chi^2 = 9.68$, $df = 1$, $p < 0.001$ Fig. 2.1b). *Glaucostegus typus* was most frequently caught in intertidal habitats (70.0 %, $n = 28$), whereas *Rhynchobatus* spp. dominated the catch in inshore coastal habitats (63.5 %, $n = 40$).

Size distribution of shark-like batoids between fishing gears and habitats

Gill-net caught *Glaucostegus typus* ranged in size from 450 to 1350 mm, *Rhynchobatus* spp. ranged from 680 to 2140 mm and *Anoxypristis cuspidata* ranged from 450 to 2430 mm (Fig. 2.2). Comparison of gill-net caught shark-like batoids size ranges between habitats was not possible due to limited sample size. Despite infrequency of capture

within inshore coastal habitat, the smallest and largest size classes of *A. cuspidata* and *Rhynchobatus* spp.were caught in this habitat.

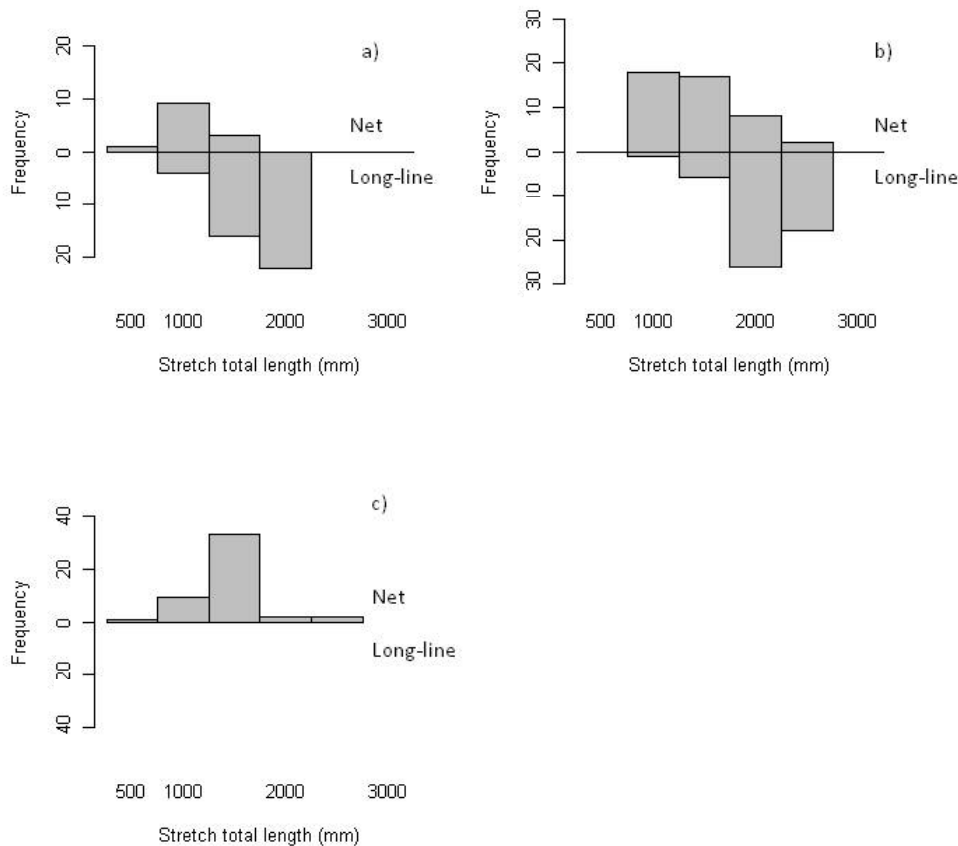


Fig. 2.2 Size distribution of shark-like batoid species in gill-net and long-line: a) *Glaucostegus typus*; b) *Rhynchobatus* spp.; c) *Anoxypristis cuspidata*. Note differences in axis scales.

Glaucostegus typus caught by gill-nets were significantly smaller (mean STL = 795 mm) than long-line caught individuals (mean STL = 2067 mm) (KS two-sample test, $D = 0.94$, $p < 0.001$, Fig. 2.2a). Long-line caught *G. typus* were smaller in intertidal habitat (range 1030 – 2700 mm, mean = 1949 mm) in comparison to inshore coastal habitat (range 1790 – 2760 mm, mean = 2232 mm), however this difference was statistically insignificant (two sample KS test, $D = 0.37$, $p = 0.08$). *Rhynchobatus* spp. caught by gill-nets (mean STL = 1213 mm) were significantly smaller than individuals caught on long-lines (mean STL = 1797 mm) (two sample KS test, $D = 0.69$, $p < 0.000$; Fig. 2.2b). Individuals caught in intertidal habitats were a similar size (range 1400-2120 mm, mean = 1791 mm) to those from inshore coastal habitats (800 – 2310 mm, mean = 1845 mm) (two sample KS test, $D = 0.20$, $p = 0.86$).

Probability of capture – gear, habitat, season

Results of GLMs indicated the probability of capturing *Glaucostegus typus* in gill-nets was affected by season, with higher probability of capture during autumn and winter. Habitat did not affect probability of capture and there was no interaction between factors (Table 2, Fig 3). *Rhynchobatus* spp. was less likely to be caught in intertidal habitats than inshore coastal (Table 2.2, Fig. 2.3). *Rhynchobatus* spp. were most likely to be caught in gill-nets during summer and winter. Although habitat type and season affected the probability of capturing *Rhynchobatus* spp. there was no significant interaction between these factors (Table 2, Fig 3). *Anoxypristis cuspidata* capture probability in gill-nets was affected by habitat but not season and there was no interaction of these factors (Table 2.2, Fig. 2.3).

The probability of capture for *Glaucostegus typus* on long-lines was not affected by season or habitat and there was no significant interaction between factors (Table 2.3, Fig. 2.3). The probability of capturing *Rhynchobatus* spp. on long-lines was significantly lower in intertidal than inshore coastal habitats. Season did not affect likelihood of capturing *Rhynchobatus* spp. Similar to gill-nets there was no interaction between habitat and season for long-line caught *Rhynchobatus* spp. (Table 2.2, Fig. 2.3).

Table 2.2 Summary of GLM analysis of shark-like batoid catch probability by gill-nets.

Factor	<i>Glaucostegus typus</i>	<i>Anoxypristis cuspidata</i>	<i>Rhynchobatus</i> spp.
Habitat	Dev = 4.43, df = 2, $p > 0.05$	Dev = 11.66, df = 2, $p < 0.05$	Dev = 18.12, df = 2, $p < 0.01$
Season	Dev = 18.09, df = 9, $p < 0.05$	Dev = 11.32, df = 9, $p > 0.05$	Dev = 28.64, df = 9, $p < 0.01$
Interaction	Dev = 1.85, df = 7, $p > 0.05$	Dev = 10.08, df = 7, $p > 0.05$	Dev = 4.74 df = 7, $p > 0.05$

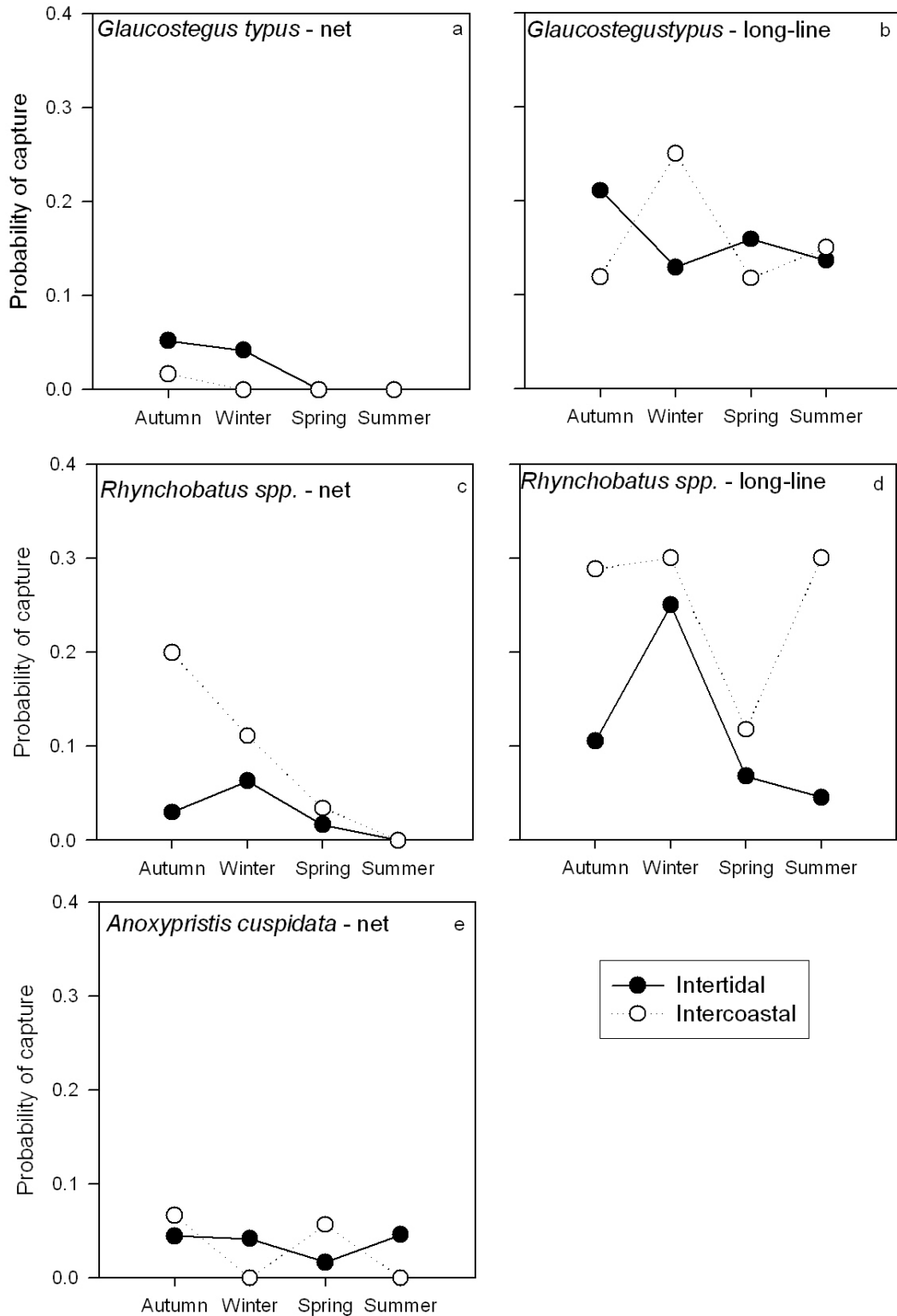


Fig. 2.3 Effect of gear, habitat and season on probability of capturing shark-like batoid species in gill-net: a) *Glaucostegus typus*, c) *Rhynchobatus spp.*, e) *Anoxypristis cuspidata*; and long-line: b) *Glaucostegus typus*, and d) *Rhynchobatus spp.*

Table 2.3 Summary of GLM analysis for shark-like batoid catch probability by long-line sampling

Factor	<i>Glaucostegus typus</i>	<i>Rhynchobatus</i> spp.
Habitat	Dev = 0.2, df = 1, $p > 0.05$	Dev = 16.42, df = 1 $p < 0.001$
Season	Dev = 0.41, df = 4, $p > 0.05$	Dev = 7.88, df = 4 $p > 0.05$
Interaction	Dev = 6.00, df = 3, $p > 0.05$	Dev = 2.44, df = 1, $p > 0.05$

2.4 Discussion

Glaucostegus typus, *Rhynchobatus* spp. and *Anoxypristis cuspidata* were all captured by gill-nets, revealing that all are available to this gear at some level. The incidence of these three species within Australian fisheries is significant due to our lack of knowledge of the species biology, movement patterns and data indicating these species are heavily harvested in adjacent regions (i.e., South-East Asia). If Australian and Asian populations of these species are linked through movement harvesting in both locations could compound mortality rates and population stability. The results of this research on how they interact with fishing gear will provide useful information for the improved management of these species throughout their range.

The size distribution of *G. typus* caught in gill-net gear was skewed toward smaller size classes that represent juvenile and sub-adult individuals based on estimates of size at maturity (Last and Stevens 2009). *Rhynchobatus* spp. caught in the gill-net gear had a wide size range (689 – 2140 mm STL) similar to catch in other Australian fisheries (Salini et al. 2007), but mortality was confined to smaller predominantly immature individuals. In comparison, individuals of both species caught on long-lines were skewed toward larger size classes. Under current fishing practices in the ECIFF these species may exist in a gauntlet fishery where fishing mortality is restricted to juvenile age classes (Prince 2005). Concentrating a fishery on a few juvenile year-classes has proven to be a robust management strategy for some elasmobranch fisheries (e.g. gummy shark, *Mustelus antarcticus*), particularly for species with low levels of productivity (Simpfendorfer 1999c, Prince 2005, McAuley et al. 2007b, Kinney and Simpfendorfer 2009). The small mesh

size of gill-nets currently used in the ECIFF limits the interaction of the ECIFF to smaller *G. typus* and is likely a robust example of the usefulness of mesh size restrictions in managing harvest. If the ECIFF included both gill-net and long-line gear types, the resulting higher catch rates and the ability to land larger mature individuals would pose a greater impact on the reproductive potential of these populations. In regions such as Indonesia where gill-nets of larger mesh sizes (> 20 cm) are utilized (W. White 2010 pers. comm. 18 May), larger size classes of shark-like batoids are caught (White and Dharmadi 2007) and population declines have been observed (White and McAuley 2003b).

Anoxypristis cuspidata was the most frequently caught shark-like batoid by gill-nets. The high catch availability of *A. cuspidata* to gill-nets can be attributed to the presence of the rostrum, which easily entangles in gill-net mesh (Simpfendorfer 2000). Given the high probability of entanglement of sawfish it is likely that most individuals that interacted with gill-nets were retained by them. The mean size of *A. cuspidata* was smaller in intertidal than inshore coastal habitats, which may indicate larger numbers of these individuals are present in this habitat. Intertidal areas may be critical habitat for early life history stages of *A. cuspidata*, potentially being used as nursery areas, foraging grounds, in predator avoidance or all of these to some extent similar to that reported for *Pristis pectinata* in the United States (Seitz and Poulakis 2002, Poulakis and Seitz 2004, Wiley and Simpfendorfer 2007, Simpfendorfer et al. 2010). When larger *A. cuspidata* were caught in nearshore waters they were sexually mature and included females carrying full term embryos (Peverell 2005). This observation suggests, these habitats are utilised by sawfish for pupping and possibly as nursery grounds, and thus may represent important areas for the implementation of conservation management.

Despite evidence from gill-net sampling that *Anoxypristis cuspidata* was present in areas where long-line sampling occurred, none were caught and thus comparisons between gears was not possible. Susceptibility to recreational hook and line and commercial long-lines has been documented for other species of sawfish (Nelson 1994, Bentley 1996b, Peverell 2005, Simpfendorfer et al. 2010) and it is unclear why *A. cuspidata* was not encountered during long-line sampling. The majority of *A. cuspidata* caught in gill-nets were less than 1500 mm, which is toward the lower size limit of *Glaucostegus typus* and *Rhynchobatus* spp. caught on long-lines. It is possible that the hooks utilized in the fisheries independent long-line gear were too large and excluded *A. cuspidata* from capture. Absence from fisheries independent sampling therefore may have been a function of gear selectivity associated with hook size; alternatively *A. cuspidata* may not readily take baited hooks.

With the exception of *Rhynchobatus* spp., it would appear that neither habitat nor season affect probability of fisheries interacting with shark-like batoids. Given these results, spatial management of fisheries activity may not provide adequate protection from fishing mortality for shark-like batoids. However, the definition of habitat in this study was coarse and further work is required to investigate preferences of habitat use at greater resolution. Similarly, the efficacy of seasonal fishing closures to provide protection may be limited given the weak association of season on catch probability for *Glaucostegus typus* and *Anoxypristis cuspidata*.

Current Management

The occurrence of shark-like batoids in mixed species fisheries such as the ECIFF is of concern, as elasmobranch catches are often poorly reported (Bonfil 1994), which makes quantifying fishing mortality difficult. Where elasmobranch species compose only a small portion of the catch, fishery activity often continues long after their collapse (Graham et al. 2001). In light of these concerns, recent management changes to the ECIFF require species specific recording of all elasmobranchs. Further, current mesh size restrictions limit fisheries interaction of the ECIFF with *Glaucostegus typus* to juveniles. Theoretically this is a desirable tactic provided fishing mortality is within the productivity limits of the species. Long-line data revealed the availability of *Rhynchobatus* spp. was highest in inshore coastal habitats and this was reflected by increased catch probabilities. Despite being morphologically similar, *Rhynchobatus* spp. and *G. typus* were not caught in equal numbers or comparable size classes by gill-net gear. Differences in catch rate may be a function of the heavier gill-nets used in regions where *Rhynchobatus* spp. are more common than *G. typus*. Similar catch probabilities for *Rhynchobatus* spp. between habitats on long-line gear further support this conclusion, since catch probabilities would be similar if individuals were available equally in both habitats.

In addition to mesh size restrictions *Anoxypristis cuspidata* is a no-take species and bag limits have been imposed for *Rhynchobatus* spp. However, bag limits imposed to reduce fishing pressure may result in product sorting whereby smaller individuals retained by the fisher are disposed of so that larger, more valuable individuals can be marketed. Under these circumstances, cryptic mortality would rise, reducing the ability to accurately assess impacts of fishing activity. In addition, previous assessments of fisheries susceptibility for *Rhynchobatus* spp. were made for a single species (*Rhynchobatus australiae*) (Stobutzki et al. 2002, Salini et al. 2007). The identification of *R. laevis* and *R. palpebratus* means that population size may have been over estimated and therefore the effect of fishing on individual species within this complex, may have been underestimated. Furthermore,

without knowledge of size-at-maturity for species within the complex it is not possible to ascertain how much of the breeding stock of each species is currently available to the fishery.

The classification of *Anoxypristis cuspidata* as a no-take species in the ECIFF may result in cryptic mortality with fishers unwilling to record negative interactions. In addition, reports from observer programs in Western Australian gill-net fisheries suggest that post-release survival of *A. cuspidata* is very low (Rory McAuley, WA Fisheries, pers. comm.). If post-release survival is low, then the designation of *A. cuspidata* as no-take will have little effect in reducing fishing related mortality, limiting efforts to quantify population stability.

The biological productivity of *A. cuspidata* is however relatively high (Tobin et al. 2010), which may explain its persistence on the Australian east coast while three co-occurring sawfish species (*Pristis microdon*, *P. clavata*, *P. zijsron*) have suffered significant reductions in range (Peeverell 2005).

Conclusion

Insufficient data regarding biology of bycatch species and their availability and incidence in fisheries is a significant hindrance to assessing a population's viability under existing fishing regimes, especially for elasmobranchs (Frisk et al. 2001). It is unclear what the ecological impact of fishing mortality sustained at the rate observed in this study will be on the long-term stability of shark-like batoid populations. Although the present study includes one of the most extensive observer survey efforts of the ECIFF to date, it represents a small proportion of the total fishing effort and consequently total fishing mortality currently experienced by shark-like batoids within the GBRWHA. Amid fears for the long-term stability of these populations semi-quantitative risk assessments have been conducted to assess susceptibility to current fishing regimes in Australian waters (Stobutzki et al. 2002, Salini et al. 2007, Zhou and Griffiths 2008). However, in the absence of information detailing life history characteristics, demographic population modelling, and taxonomic resolution in species identification of *Rhynchobatus* spp., the true impact of fisheries mortality is difficult to quantify. A clear understanding of the biology, ecology, species status and harvest by fisheries will be key to effective conservation of these species to halt any further decline of their populations.

CHAPTER 3

Application of baited remote underwater video surveys to quantifying elasmobranch spatial distribution at an ecosystem scale



Plate 3. *Rhynchobatus* sp. approaching a BRUVS[®] on the Great Barrier Reef. Photo taken by Mike Cappo (Queensland, 2008).

3.1 Introduction

Understanding how a species is distributed within an ecosystem is important to conservation and management planning (Colton and Swearer 2010, Brooks et al. 2011). Quantitative information describing species distribution allows assessment of overlap with threats such as fishing, pollution and habitat loss. Management responses are likely to be more effective when explicit distributional information for a target species is available. Mounting pressure from fishing (Bonfil 1994), habitat degradation (Jennings and Kaiser 1998, Jennings et al. 2008), pollution (Gelsleichter et al. 2005) and climate change (Chin et al. 2010) has seen the decline of some of the world's elasmobranch populations (Dulvy et al. 2008). The mobility of most elasmobranch species presents significant challenges to their assessment and management (Knip et al. 2012a). Species ranges may extend across jurisdictional borders, with the critical habitat of species occurring in regions with differing approaches to environmental regulation and management. Understanding species distributions is thus important to rebuilding depleted populations and stabilizing populations of species harvested as primary targets or as a bycatch.

Naturally low abundances, broad geographic distributions, low economic value and poor taxonomic resolution of fisheries records have all contributed to a state of data deficiency for elasmobranch species (Walker 2004, Dulvy et al. 2008, Lack and Sant 2008, Dulvy and Forrest 2010, Simpfendorfer et al. 2011a). Knowledge of elasmobranch species distribution and status has traditionally come from fisheries-dependent data in the form of catch and effort statistics derived from fishery logbooks (Crow et al. 1996, Stobutzki et al. 2002, Walsh et al. 2002, Baum et al. 2003) and observer surveys (Stevens 1992, Marin et al. 1998, Burgess et al. 2005, Harry et al. 2011b). However, bias associated with gear selectivity between life history stages and non-random distribution of fishing effort limit the application of these data to describing a species' spatial distribution. Fisheries-independent sampling tends to employ commercially used gears (e.g. gill-net, trawl, seine, long-line) in conjunction with a more robust sample design to survey community diversity and species distribution (Andrew et al. 1997, Simpfendorfer et al. 2002b, Walker et al. 2005), but employment of the same gears as fisheries-dependent sampling constrain the data through gear selectivity (Simpfendorfer and Unsworth 1998, McAuley et al. 2007a). Furthermore, sampling tends to be confined to areas where target species are likely to be encountered. Fisheries-independent sampling programs may also be limited by where certain gears can operate. Broad scale exploratory sampling is seldom conducted due to limitations of both time and money. In addition, sampling in defined

geographic locations may not provide accurate estimates of species distribution, particularly if normal movements of a species are large relative to the spatial distribution of the survey (Rago 2004). Targeted sampling to investigate species presence in an area may also fail to encompass variations in seasonal migration or foraging patterns (Rago 2004). Thus traditional sampling programs may not always provide the best data to define species distributions.

Non-extractive survey techniques have been applied to site attached species that are easily enumerated (Harvey et al. 2002), but these techniques may be less effective for more mobile species (Ward-Paige et al. 2010). Diver based underwater visual survey (UVS) have been used to determine the abundance and distribution of elasmobranch species (Edgar et al. 2004, Castro and Rosa 2005, Robbins et al. 2006). However, short survey times, mobility of target species (Sale and Douglas 1981, Kulbicki 1998, Ward-Paige et al. 2010), behavioural responses to divers (Kulbicki 1998, Watson and Harvey 2007), depth, and visibility limitations of SCUBA all impede the ability of UVS to fully quantify elasmobranch abundance and distribution. As a result, UVS tends to produce biased population density estimates (Sale and Sharp 1983, Edgar et al. 2004), particularly when surveying large mobile species such as elasmobranchs (Ward-Paige et al. 2010).

Baited remote under-water video station (BRUVS[®]) offer a standardized, non-extractive technique for assessing species diversity and habitat presence across geographically wide areas, depth ranges and varied habitats (Cappo et al. 2004). Large elasmobranchs have been sighted commonly in BRUVS[®] footage (Meekan et al. 2006, Malcolm et al. 2007, Brooks et al. 2011). The passive nature of this methodology allows for its application in fragile and protected areas, and when dealing with rare and threatened species that may be negatively affected by intrusive gears or capture. Utilisation of BRUVS[®] avoids problems of mesh and hook selectivity encountered when using extractive sampling techniques, and some of the biases of behavioral avoidance associated with UVS. However, this technology is not without its own biases, including: attraction to bait, avoidance by prey species due to attraction of predators, reduced effectiveness in low light and high turbidity conditions and the potential to repeatedly count the same individuals in subsequent surveys (Langlois et al. 2006). Despite this, the data generated by BRUVS[®] provides a permanent record allowing impartial, repeatable measurements and enabling standardized data collection (Cappo et al. 2004, Cappo et al. 2007). While BRUVS[®] have been used to compare the abundance of sharks between areas (Meekan et al. 2006, Brooks et al. 2011) and investigate prey communities (Meekan and Cappo 2004, Gutteridge et al. 2011), their utility to determine the distribution of elasmobranchs at broad spatial scales and across a range of habitat types remains

untested. The present study examined the distribution of shark-like batoids (families Rhynchobatidae, Rhinobatidae and Pristidae) at broad spatial scales within the Great Barrier Reef World Heritage Area. The specific aims were: (1) examine the utility of BRUVS[®] to determine shark-like batoid distribution, and (2) examine factors that define the distribution of this group of species.

3.2 Materials and Methods

Study species

The giant shovelnose ray *Glaucostegus typus*, whitespotted guitarfish *Rhynchobatus* spp., shark ray *Rhina ancylostoma*, narrow sawfish *Anoxypristis cuspidata*, and green sawfish *Pristis zijsron* are shark-like batoids that have wide geographic distributions and are globally threatened in International Union for Conservation of Nature (IUCN) assessments (McAuley and Compagno 2003a, b, White and McAuley 2003a, b, Compagno et al. 2006b). Despite their capture in commercial fisheries within the Great Barrier Reef World Heritage Area (Harry et al. 2011b), there is limited understanding of how shark-like batoids are distributed within their geographic ranges. *Glaucostegus typus* is listed as 'Vulnerable' on the IUCN Red List of Threatened Species, based on intensive fishing pressure and population declines in South-East Asia (White and McAuley 2003a). Although fishing effort in Australia is not as intense as South-East Asia, *G. typus* has been classified as 'high risk' in ecological risk analyses due to distributional overlap with multiple fisheries (notably gill-net and prawn trawl) and low productivity (Salini et al. 2007). Within Australia, the Family Rhynchobatidae consists of three distinct species (*Rhynchobatus australiae*, *R. laevis* and *R. palpebratus*), which have been consistently confused in the literature (Last and Stevens 2009). Although each of the species within the complex is listed as 'Vulnerable' on IUCN Red List of Threatened Species the extent of individual species decline and range reductions are hard to quantify given the taxonomic confusion (McAuley and Compagno 2003b, White and McAuley 2003a, Compagno and Marshall 2006). Current management strategies within Queensland waters treat the species complex as a single group due to difficulties in identifying species. Thus, I have treated all individuals as a group that will herein be referred to as *Rhynchobatus* spp. Very little is known about *R. ancylostoma*. Fisheries data suggests it is widely distributed throughout the Indo-West Pacific, however, it does not appear to be common (McAuley and Compagno 2003a). *Rhina ancylostoma* is assessed globally as 'Vulnerable' on the IUCN Red List of Threatened Species due to evidence of population decline, occurrence in multiple fisheries and the high value of its fins (McAuley and Compagno 2003a). However, given the paucity of data it is unclear whether this

assessment accurately categorizes the threat faced by *R. ancylostoma*. Similar to other sawfish species (Simpfendorfer 2000, Cavanagh et al. 2003, Carlson et al. 2007), *A. cuspidata* and *P. zijpsron* have suffered substantial reductions in abundance with populations now fragmented throughout their range (Compagno et al. 2006a, Compagno et al. 2006b). Consequently, both are listed as 'Critically Endangered' on the IUCN Red List of Threatened Species (Compagno et al. 2006a, Compagno et al. 2006b).

Field methods

A series of baited remote underwater video stations (BRUVS[®]) were deployed in lagoonal and inter-reef waters of the Great Barrier Reef Marine Park (GBRMP) between March 2000 and May 2010 to survey vertebrate communities (Cappo et al. 2004, Cappo et al. 2007). The BRUVS configuration consisted of a galvanized, trestle-shaped frame, enclosing a simple camera housing made from PVC pipe with acrylic front and rear ports. Sony Mini-DV HandiCams with wide-angle lens adapters (0.6×) were used in housings. Exposure was set to 'Auto', focus was set to 'Infinity/Manual', and 'Standard Play' mode was selected on the cameras. Detachable bait arms (20 mm plastic conduit) had a 350 mm plastic mesh canister containing 1 kg of crushed sardines (*Sardinops* or *Sardinella* spp.) as bait, lying on the seabed. BRUVS units were deployed with 8 mm polypropylene ropes and polystyrene surface floats bearing a marker flag and were retrieved with a hydraulic pot-hauler. The BRUVS were deployed to provide between 45 - 90 minutes of video imagery recorded at the seabed (Cappo et al. 2004, Cappo et al. 2007).

A total of 2471 BRUVS[®] deployments were conducted between latitudes 10.7 ° S and 24.2 ° S, and longitudes 143.38 ° E and 152.36 ° E (Fig. 1). BRUVS[®] were deployed in a depth range of 7 -115 m, with a mean sampling depth of 36 ± 15.6 m. Some deployments were shortened due to loss of bait, toppling of the BRUVS[®] by currents, or other factors. The mean length of video imagery was 59.81 ± 9.18 minutes. Only 5 % of all deployments exceeded the median of 62.42 minutes, and 10 % were less than 50 minutes.

Data analysis

Interrogation of each tape was conducted using a custom interface (BRUVS[®] 1.5.mdb, Australian Institute of Marine Science, 2006) to manage data from field operations and tape reading, to capture the timing of events, and to capture reference images of the seafloor and fish in the field of view. Shark-like batoids observed on tapes were identified to species.

For each species, records were made for each species of shark-like batoid of the maximum number of individuals seen together at any one time on the whole tape (MaxN). The use of MaxN as an estimator of relative abundance has been reviewed in detail by Cappo et al. (2004).

Depth and latitude in which shark-like batoids were observed on BRUVS[®] was compared to all depths and latitudes at which BRUVS[®] were deployed using Chesson's α (Chesson 1978):

$$\alpha = (r_i / p_i) / \sum (r_i / p_i)$$

where r_i was the proportion of time a shark-like batoids spent in depth or latitude i , and p_i was the proportion of BRUVS[®] at depth/latitude i . The value of α can range from 0 (strong avoidance) to 1 (strong affinity), with values $> (1/\text{number of categories})$ indicating electivity and values $< (1/\text{number of categories})$ indicating avoidance.

Chi squared (χ^2) contingency tables were used to compare proportions of *Rhynchobatus* spp. occurring in BRUVS[®] between seasons and fished to no-take areas. Zonation within the Great Barrier Reef Marine Park was changed as part of the representative areas program (RAP) in 2004 (Fernades et al. 2005), which was during the BRUVS[®] sampling period. Therefore pre RAP zonation was used to define fished and no-take area in the present study. Species density and abundance in marine systems typically change with latitude driven by changes in environmental condition (Travers et al. 2012). Community structure is also affected by distance from shore due to species occupying shallow or deeper habitats with preferences driven by variation in abiotic conditions (Travers et al. 2012). Thus depth was used as categorical variable to assess the effect of distance from shore rather than longitude (Connell and Lincoln-Smith 1999) as bins of longitude would not allow for meaningful comparisons. Latitude was pooled into six bins to create a categorical variable for analysis. Bins representing 2.5 degrees of latitude were considered the optimal size. *Glaucostegus typus* and *R. ancylostoma* were excluded from chi squared (χ^2) contingency tables due to limited sample size.

The biological and physical diversity of Great Barrier Reef World Heritage Area (GBRWHA) is divided into 70 bioregions (http://www.soe-townsville.org/data/coastal/marine-bioreg/bioregions_description.pdf). Each bioregion was categorized according to bathymetry, substrate and biology of the area (Fernades et al. 2005). BRUVS[®] were conducted in forty-one of the 70 bioregions within the GBRMP. The substrate type categorical variable (e.g. fine or coarse sediment, complex rocky, coral, foraminifera/*Halimeda* deposits, gravel, hard substrate seafloor, high carbonate sand,

mud, reef, sand, shelly-coarse sediment and terrigenous mud) was taken from the description for each bioregion. There are seven management zonations in the GBRMP designed to regulate activity and conserve biodiversity (Table 1). For the purposes of this analysis, zone type (7 factors) and fished/unfished (2 factors) were used. Months of sampling were collated into nominal seasons: summer (December - February), autumn: (March - April), winter (June - August) and spring (September - November) for analysis. The effect of location (latitude/longitude), distance along reef, distance across shelf, substrate type, marine park zone, water depth (continuous) and season on the probability of encountering *Rhynchobatus* spp. was estimated using a logistic generalised linear model (GLM) with a binomial error structure and logit link function. The GLM only considered first order interactions due to limitations in the data. Since the number of individuals observed at individual BRUVS was low, only presence/absence was considered.

The occurrence (presence) of *Rhynchobatus* spp. was analysed using spatial position across and along the GBRMP, and depth, as explanatory variables. Introduced by Fabricius & De'ath (2001), the theory behind the use of these "heuristic" spatial predictors was that any sampling region encompassing environmental gradients can be spatially divided in three dimensions in a manner that represents those gradients. Furthermore, it acknowledges that many mechanistic environmental factors can vary along the same gradient, and it is probable that some of these may be important but not measured (or measurable) in a given sampling program. This can lead to spurious inferences about the variables that have been measured. Cross-shelf position was set to 0 on the coast and 1 on the 80 metre isobath on the shelf edge. The along-shelf position was set to 0 on the south west limit of the GBRMP and 1 on the north eastern limit. The corners of the polygon formed in this way were 142.530° E, -10.690° S; 144.060° E, -10.680° S at the northern end, and 152.490° E, -25.000° S; 152.900° E, -24.220° S at the southern end.

3.3 Results

Shark-like batoid presence

One hundred and nineteen shark-like batoids were recorded during 2471 BRUVS[®] deployments (Fig 3.1). *Rhynchobatus* spp. were the most frequently sighted and widely distributed of the shark-like batoids, accounting for 105 (88.3 %) of total encounters and occurring across all 12° of latitude. Nine *Rhina ancylostoma* and five *Glaucostegus typus* were also present. On two occasions *Rhynchobatus* spp. and *G. typus* were sighted on the same BRUVS[®]. Neither sawfish species known to inhabit the GBRMP (*Anoxypristis cuspidata*, *Pristis zijsron*) was observed.

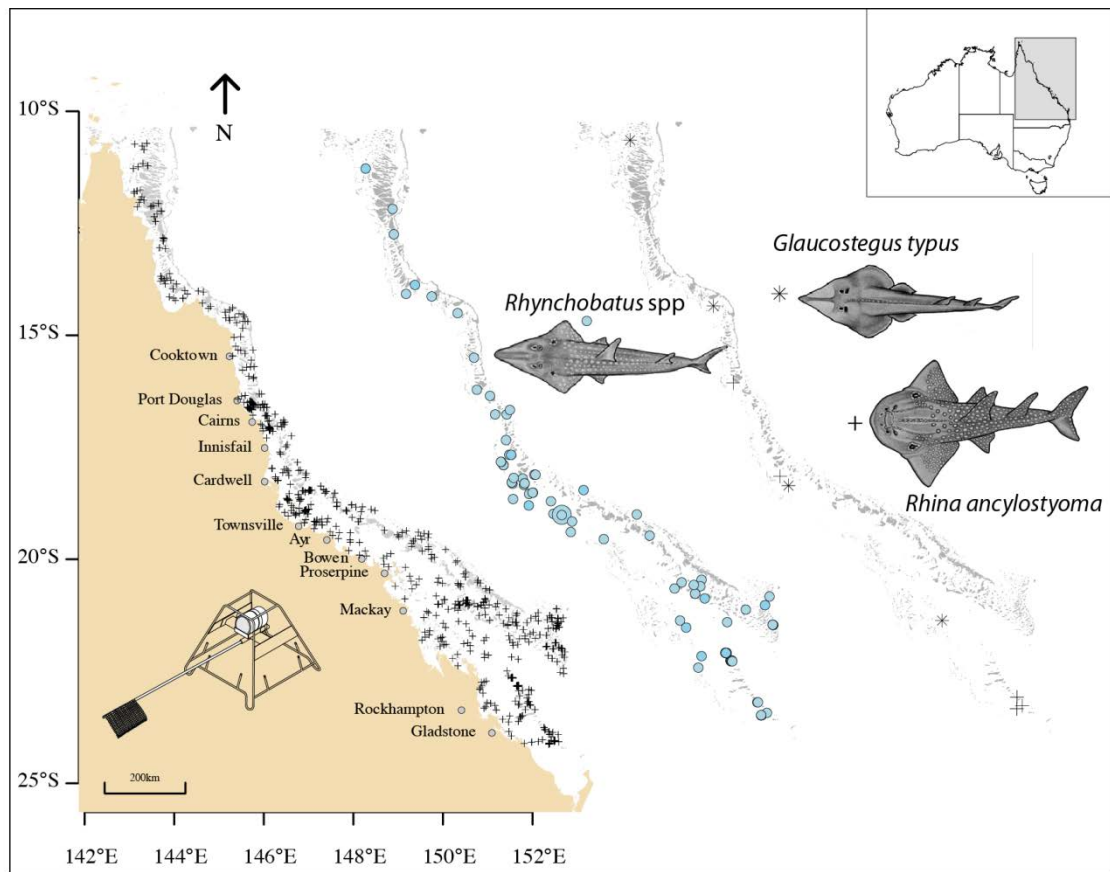


Fig 3.1 Location of all 2471 BRUVS[®] sets within the Great Barrier Reef Marine Park, location of sightings of *Rhynchobatus* spp. = ●, * = *G. typus*, + = *R. ancylostoma*). All symbols are scaled to a unit of abundance.

Table 3.1 Summary of access and use for each of the current marine park zones within the Great Barrier Reef World Heritage Area. Shading indicates zones open to commercial and recreational fishing.

Zone	Fishing	Recreational Fishing	Commercial Fishing	Access	Tourism
Preservation	No-take	No	No	No	No
Marine National Park	No-take	No	No	Yes	Yes
Scientific Research	No-take	No	No	No	No
Buffer	Fished	Yes	No	Yes	Yes
Conservation Park	Fished	Yes	No	Yes	Yes
General Use	Fished	Yes	Yes	Yes	Yes
Habitat Protection	Fished	Yes	Yes	Yes	Yes

There was no apparent relationship between soak time of the BRUVS and probability of sighting a shark-like batoid. Only two (both *Rhynchobatus* spp.) were sighted after 62.5 minutes had elapsed. The range and means (\pm standard deviations) of the number of minute's elapsed to first sighting were: for *R. ancylostoma* 9.15-52.5 (30.9 \pm 15.3), for *Rhynchobatus* spp. 5.4-80.1 (34.8 \pm 16.7), and 5.2-44.7 (31.9 \pm 16.6) for *G. typus*.

Affinity for depth or latitude

Shark-like batoids were present on BRUVS[®] in depths between 10.4 and 87.5 m. Shark-like batoids had the highest affinity for depths from 30 to 40 m (Fig 3.2a). Sampling in this depth ranges accounted for 56.7 % of total BRUVS[®] effort. Shallow depths (7 - 20 m; 38.3 % of BRUVS[®]) and those greater than 70 m (3.6 % of BRUVS[®]) had fewer sightings. Shark-like batoids were present throughout the latitudinal range (Fig 3.2b). Fifty five percent of total sampling occurred across five degrees of latitude (17.5 to 22.5^o) and accounted for 87.4 % of shark-like batoid sightings.

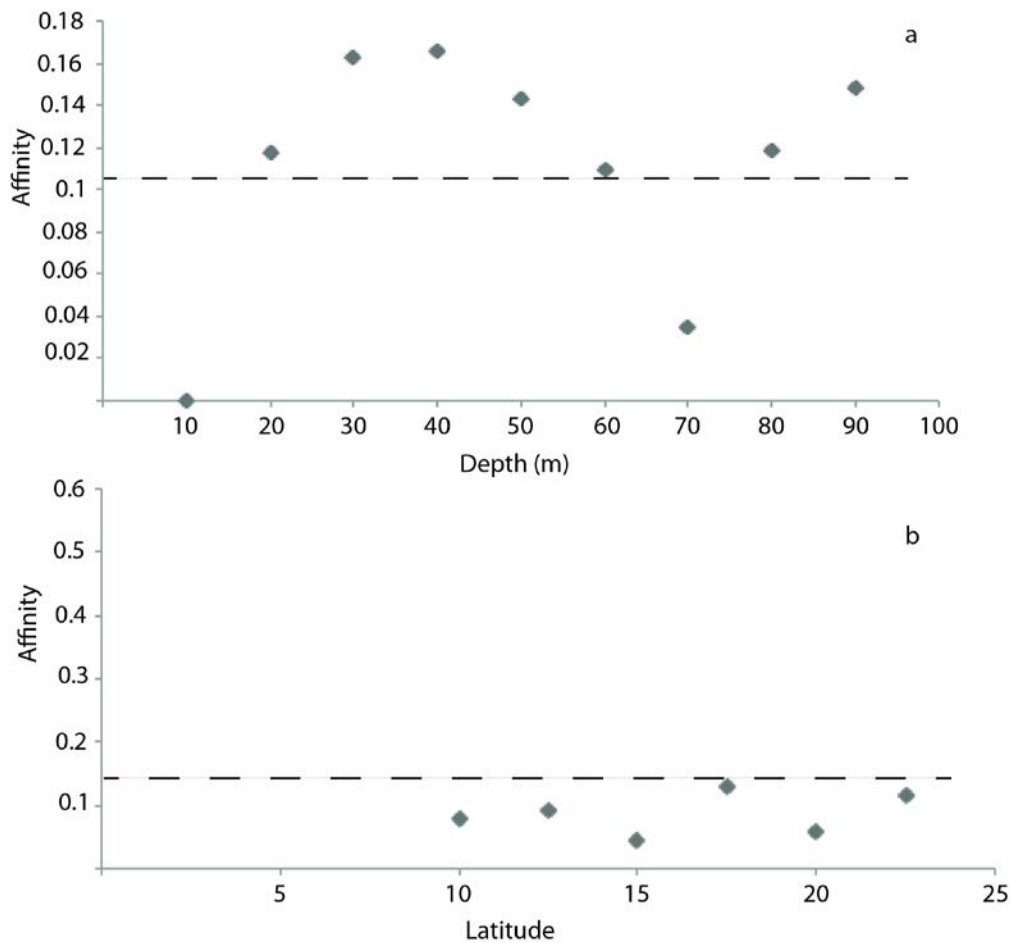


Fig 3.2 Affinity of shark-like batoids across (a) depth and (b) latitude ranges at which BRUVS[®] sampling occurred. Hashed line represents expected affinity.

Effect of season on presence

The proportion of *Rhynchobatus* spp. present between seasons was not significantly different ($\chi^2 = 0.60$, $df = 3$, $p = 0.90$), with individuals most commonly recorded during winter (10 of 142 BRUVS[®]; 7.0 %) and autumn (33 of 645[®]; 5.1 %), while spring (38 of 883 BRUVS[®]; 4.3 %) and summer (23 of 801 BRUVS[®]; 2.9 %) had lower occurrence.

Effect of marine park zone on presence

Rhynchobatus spp. were most frequently sighted in habitat protection (37 of 703 BRUVS[®]; 5.0 %), marine national park (38 of 814 BRUVS[®]; 4.5 %), general use (38 of 753 BRUVS[®]; 4.4 %) and conservation park (3 of 74 BRUVS[®]; 3.9 %) zones. The number of BRUVS[®] conducted in the port, buffer and preservation zones were too few to draw any

conclusions regarding the presence or preference of *Rhynchobatus* spp. for these areas. There was no difference in the proportion of *Rhynchobatus* spp. encountered between zones ($\chi^2 = 0.08$, $df = 5$, $p = 0.993$) or between fished and no-take areas ($\chi^2 = 0.004$, $df = 1$, $p = 0.948$).

Effect of bioregion on presence

Shark-like batoids were present in 20 of the 41 bioregions sampled (Table 3.2), with highest numbers observed in the NN Capricorn Bunker Banks (29 of 285 BRUVS[®]; 10.2 %), the NB5 Inner Mid Shelf Lagoon (18 of 188 BRUVS[®]; 9.6 %) and NB3 Inner shelf sea grass (12 of 209 BRUVS[®]; 5.7 %). Bioregions with the highest number of shark-like batoids present as a proportion of sampling effort were NJ Princess Charlotte Bay Outer Shelf (1 of 6 BRUVS[®]; 16.6 %), NK Princess Charlotte Bay (1 of 6 BRUVS[®]; 16.6 %), RB1 Far Northern Outer Mid Shelf Reefs (2 of 17 BRUVS[®]; 11.7 %) and NB5 Inner Mid Shelf Lagoon (18 of 188 BRUVS[®]; 9.6 %). *Rhynchobatus* spp. accounted for the majority of shark-like batoid presence in bioregions. *Rhina ancylostoma* and *G. typus* were most commonly present in NB8 Capricorn Bunker Lagoon and NB5 Inner Mid Shelf Lagoon bioregions, respectively (Table 3.2). The NB5 Inner Mid Shelf Lagoon was the only bioregion in which all three species of shark-like batoids were present.

Table 3.2 Summary of shark-like batoid encounters between bioregions including: percentage of the GBRMP that the bioregion constitutes (% GBRMP), percentage of the bioregion that is currently designated as no-take (% No-take), dominant substrate type of bioregion as specified by bioregion descriptions, and number of individuals sighted for *Rhina ancylostoma* (SRY), *Glaucostegus typus* (GSR), and *Rhynchobatus* spp. (WSG). Only bioregions in which shark-like batoids were encountered have been included.

Bioregion	Bio region area (km ²)	% GBRMP	% No-take	Substrate	No. BRUVS [®]	SR Y	GS R	WS G
NA3 High Nutrients Coastal Strip	17,154	5	<1	Mud	145			5
NB1 Inshore Muddy Lagoon	8,889	3	25	Mud	39			1
NB3 Inner Shelf Seagrass	7,466	2	<1	Mud	209	1		11
NB5 Inner Mid Shelf Lagoon	9,836	3	<1	Sand	188	3	2	14
NB6 Inner Shelf Lagoon Continental Islands	14,619	4	<1	Mud	63		1	2
NB7 Mid Shelf Lagoon	25,112	7	<1	Mud	98			3
NB8 Capricorn Bunker Lagoon	16,647	5	<1	Seagrass	305	3		7
NC Mid Shelf Inter Reef - Seagrass	5,515	2	17	Mud	34		1	
NJ Princess Charlotte Bay Outer Shelf	685	<1	0	Sand	6			1
NK Princess Charlotte Bay	1,440	<1	3	Mud	6			1
NL2 Outer Shelf Seagrass	5,017	1	3	Mud	108			2
NL3 Outer Shelf Inter Reef - Central	11,701	3	3	Gravel	147			6
NL4 Outer Shelf Inter Reef - Southern	17,719	5	4	Gravel	185			8
NL5 Swains Inter Reef	14,325	4	8	Sponge	246			7
NM Mid Shelf Seagrass	2,832	1	4	Mud	22		1	2
NN Capricorn Bunker Banks	2,363	1	0	Gravel	285			29
X4 Capricorn Bunker Inter Reef	1,717	<1	2	Reef	82	2		2
X5 Outer Central Inter Reef	5,033	1	1	Gravel	36			2
X7 Central Inter Reef	9,595	3	3	Gravel	42			1
RB1 Far Northern Outer Mid Shelf Reefs	1,440	<1	8	Reef	17			2

Factors affecting presence of Rhynchobatus spp.

Generalized linear modelling indicated that latitude, longitude or depth of BRUVS[®] deployments did not affect the probability of encountering *Rhynchobatus* spp. within the GBRMP (Table 3.3). Season significantly affected the probability of encountering

Rhynchobatus spp., although, in combination with other factors there were no significant interactions with season (Table 3.3). Substrate also affected the probability of encountering *Rhynchobatus* spp. with mud and gravel found to be the preferred habitat types. Despite differences in both total abundance and proportion of encounters between marine park zones GLMs indicated there was no significant effect of zone on the likelihood of encountering *Rhynchobatus* spp. (Table 3.3).

Table 3.3 Summary of GLM analysis for probabilities of encountering shark-like batoids by BRUVS[®]

	AIC	DEV	DF	P
Latitude	945.8	0.45	1	0.45
Longitude	946.1	0.02	1	0.86
Across	942.7	3.44	1	0.06
Along	946.0	0.11	1	0.73
Depth	944.7	1.36	1	0.24
Season	941.0	9.11	3	0.02
Zone	953.1	2.97	6	0.81
Substrate	1020.5	41.31	16	0.00
Season* depth	944.8	1.88	3	0.59
Season* latitude	945.6	3.11	3	0.37
Season* longitude	944.1	3.27	3	0.35



Fig 3.3 Images of shark-like batoid encounters from BRUVS. *Rhynchobatus* spp. (a,b). *Glaucostegus typus* (c), *Rhina ancylostoma* (d).

3.4 Discussion

This study demonstrates that BRUVS[®] can be a useful tool in examining the distribution and habitat associations of large mobile elasmobranchs at broad spatial scales. The use of BRUVS[®] provided a non-invasive, non-destructive and minimally disruptive approach that is appropriate for species of conservation interest. Given the naturally low abundances of shark-like batoids, ecological sensitivity and structural complexity of some habitats within the GBRMP, the use of conventional fisheries independent sampling across such a large geographic area would be impossible. Fisheries-dependent data for this group is restricted by regulation of the fishing industry both spatially and temporally. Utilisation of BRUVS[®] avoided these issues and provided information on the distribution, seasonality and habitat use of *Rhynchobatus* spp. Application of BRUVS[®] technology to define shark-like batoid distribution extends beyond previous use to assess species diversity and compare abundance in isolated locations (Meekan et al. 2006, Brooks et al. 2011).

Rhynchobatus spp. were encountered across a number of bioregions, with highest incidence on the inner and mid-shelf regions of the GBRMP. Within these bioregions the

commonality was substrate type dominated by a mud and gravel composition. BRUVS[®] data suggest *Rhynchobatus* spp. are habitat generalists, occurring in multiple regions and thus habitat types across the GBRMP. However, this taxa is actually comprised of a complex of three species, which may have different distributions. Thus further research that employs techniques allowing for better discrimination of these species will be required before species-specific information is available.

Seasonal changes in habitat use are well documented for elasmobranch species (Hunter et al. 2006a, Weng et al. 2007, Carlisle and Starr 2009) and are often associated with changes in water temperature (Casey and Kohler 1992, Heithaus 2001), salinity (Heupel and Simpfendorfer 2008, Ubeda et al. 2008, Knip et al. 2011b) or philopatric behaviour associated with mating/pupping (Colman 1997, Gunn et al. 1999, Hueter et al. 2004). Higher probability of encountering *Rhynchobatus* spp. during cooler months suggests a seasonal pattern in habitat use. Consistent with the present study *Rhynchobatus laevis* (one member of the *Rhynchobatus* spp. complex) was among a suite of ten elasmobranch species observed to move into inshore waters of Shark Bay, Western Australia during warmer months (Vaudo and Heithaus 2009). It was suggested this movement was a behavioural trait to exploit local thermal heterogeneity for physiological gains (Vaudo and Heithaus 2009). Movement of *Rhynchobatus* spp. into inshore areas of the GBRMP is likely linked to several factors (e.g. foraging, physiological gains and reproduction) that combine to create a net gain from use of these areas rather than a single driver. The use of inshore areas by *Rhynchobatus* spp. requires further attention to elucidate how these areas are used, and which habitats are preferred to further quantify the seasonal behaviour revealed here.

The lack of difference in presence between fished and no-take areas suggests that MPAs may be of limited benefit for shark-like batoids. The success of no-take marine reserves has largely been limited to site attached species or those with high site fidelity (Murawski et al. 2000, Galal et al. 2002, Russ et al. 2004). Recent research using BRUVS[®] to quantify reef shark presence in MPAs has revealed benefits of protected zones to several shark species (Bond et al. 2012, Goetze and Fullwood 2013). However, many of the reef shark species examined are known to have high site fidelity and habitat dependence. If *Rhynchobatus* spp. are not resident within discrete areas or habitats then small-scale MPAs may not be beneficial. Similarities in encounterability of *Rhynchobatus* spp. between fished and no-take areas suggest that little refuge is offered by the current matrix of zones within the GBRMP. These results suggest limited site fidelity and broad-scale movement of individuals, but further research is required to fully address this.

Given the small number of *R. ancylostoma* and *G. typus* encountered it was difficult to identify spatial or seasonal patterns for their occurrence. Vaudo and Heithaus (2009) observed year round presence of *G. typus* in inshore coastal habitats of Shark Bay. Data from the present study supports these findings with all encounters of *G. typus* occurring in highly turbid inshore waters. Other species of Rhinobatidae (*Rhinobatos productus* and *Zapteryx brevirostris*) have also been found to prefer inshore coastal and estuary habitats (Farrugia et al. 2011). The utility of BRUVS[®] is severely limited by high turbidity so they are mostly unsuitable for quantifying spatial ecology of *G. typus* given the species preference for inshore coastal waters with muddy seabeds (White and Potter 2004, Pierce et al. 2011, White et al. 2012). *Rhina anclystoma* was also encountered in low numbers, with the majority of individuals observed in lagoonal or mid-shelf reef waters. Little is known about the spatial ecology of this species, but it is thought to inhabit coastal and reef regions preferring muddy or sandy substrate (Gordon 1992, McAuley and Compagno 2003a). BRUVS[®] may be limited in utility for very rare species such as *R. anclystoma* however, given the conservation concern and paucity of data for this species any advancement in knowledge is a valuable addition to understanding its ecology. The lack of any sightings of sawfish may be attributed to their low population size and use of highly turbid inshore habitats. Fisheries independent long-line sampling in regions where individuals were known to be present (based on capture in gill-nets) also failed to catch *A. cuspidata* (White et al. 2012). Thus *A. cuspidata* and *P. zisron* may not be attracted by baits associated with hooks or BRUVS[®].

Conclusion

The application of BRUVS[®] to quantifying spatial distribution of elasmobranchs has bridged gaps between historic fisheries records and fisheries independent data. Although BRUVS[®] can be a good tool for surveying some species (e.g. *Rhynchobatus* spp.), small population sizes and/or habitat preference for highly turbid areas make its application limited for others (e.g. *G. typus*, *R. ancylostoma*, *A. cuspidata*, *P. zisron*). However, given the paucity of data for all shark-like batoids, the observations made in BRUVS[®] surveys can provide a valuable contribution to our understanding of spatial ecology for these populations.

CHAPTER 4

Spatial ecology of shark-like batoids in a large coastal embayment



Plate 4: Surgical implantation of acoustic transmitter into a female *Glaucostegus typus*.
Photo by Jon Smart (Cleveland Bay, 2010).

4.1 Introduction

Understanding spatial ecology is essential for quantifying vulnerability to exploitation and to understanding the benefits of conservation management (Simpfendorfer et al. 2010, Farrugia et al. 2011, Simpfendorfer et al. 2011a). Thus identification of critical areas (e.g. nursery or mating areas), understanding the seasonality of their use, and vulnerability of these habitats to anthropogenic impacts, all contribute to the development of appropriate management strategies (Simpfendorfer et al. 2011a, Yates et al. 2012). If species utilise specific habitats during key life history stages or exhibit strong site fidelity, then localized impacts (e.g. fisheries and habitat alteration) could have significant consequences for populations (Knip et al. 2012c).

The current understanding of elasmobranch spatial ecology has largely come from research on shark species, and more specifically those with a fusiform body form such as the Carchariformes and Lamniformes (Conrath and Musick 2010, Heupel et al. 2010, Speed et al. 2010, Knip et al. 2011a). Despite a surge in acoustic monitoring studies (Voegeli et al. 2001, Heupel et al. 2006) and application of this approach to numerous elasmobranch species (Heupel and Webber 2012), the spatial ecology of batoids remains poorly understood (Vaudo and Heithaus 2012). One group of batoids – the shark-like batoids (i.e. families Rhynchobatidae, Rhinoabatidae, Rhinidae and Pristidae) which are morphologically similar to sharks in having an elongate body and well developed caudal fin – have been particularly poorly studied. There is little information describing shark-like batoid habitat preferences and movements and how these behaviours change with life history stage. What is known largely comes from fisheries dependent catch and effort data (White et al. 2013a) or visual surveys (Vaudo and Heithaus 2009). However, spatial regulation of fishing effort and gear selectivity, in addition to poor taxonomic resolution of bycatch species limit the utility of these data for assessing distribution and habitat preference of non-target species.

Nearshore areas provide critical habitat for elasmobranch species (Heupel et al. 2007, Knip et al. 2010) and function similarly for at least some shark-like batoids (Simpfendorfer et al. 2010). Although multiple shark species may inhabit the same nearshore region (Simpfendorfer and Milward 1993), use may be partitioned by habitat or prey community composition (White and Potter 2004, Pikitch et al. 2005, DeAngelis et al. 2008). Further, there often is a temporal component to partitioning with changes between seasons, cohorts or between life history stages (Knip et al. 2011a, b). Thus understanding the use of nearshore areas by shark-like batoids will be important for designing effective conservation strategies where they are needed.

The giant shovelnose ray, *Glaucostegus typus* and whitespotted guitarfish, *Rhynchobatus* spp. are shark-like batoids listed in threatened categories in the International Union for Conservation of Nature (IUCN) Red List assessments. Intensive fishing pressure has resulted in population declines in South-East Asia (White and McAuley 2003a). The morphology of these species has implications for their capture and retention in commercial fisheries where interactions are more akin to those of shark species than typical dorsal ventrally flattened batoid species. It is currently unclear whether the shark-like morphology of these species will also affect how they use space within an ecosystem. In Australia, *G. typus* has been classified as 'high risk' in ecological risk assessments due to distributional overlap with multiple fisheries (notably gill-net and prawn trawl) and assumed low productivity (Salini et al. 2007). The *Rhynchobatus* spp. complex in Australia is comprised of three distinct species, *Rhynchobatus australiae*, *R. laevis* and *R. palpebratus* that have consistently been confused in the literature (Last and Stevens 2009). The species complex in Australian waters has made assessing the level of threat to this group challenging. Current management strategies within Queensland waters treat the species complex as a single group due to difficulties in identification. Thus I have treated all individuals as a group that will herein be referred to as *Rhynchobatus* spp. Although fishing effort in Australia is not as intense as South-East Asia they are taken in fisheries and development is altering the habitat, hydrology and water quality of nearshore areas (U.N. 2012). If nearshore areas are critical habitat for shark-like batoids, then significant development in these regions may have long-term implications for the stability of these populations.

Fisheries dependent data suggest both juvenile and adult *G. typus* and *Rhynchobatus* spp. occur within the same nearshore areas in northern Queensland (White et al. 2013a). However, habitat utilisation by these morphologically similar species, and whether these nearshore areas represent important habitats, remains unclear. The purpose of the present study was to examine: 1) residency of two shark-like batoids within a nearshore region; 2) compare activity space size between and within species; and 3) investigate changes in spatial ecology based on size and sex within species.

4.2 Materials and Methods

Study Location

Cleveland Bay ($19^{\circ}12'3''\text{S}$, $146^{\circ}54'4''\text{E}$) is a shallow water embayment situated in the central region of the Great Barrier Reef World Heritage Area (GBRWHA) (Fig 4.1). The bay is approximately 27 km wide and covers an area of 225 km². The majority of the bay is less than 10 m deep with a maximum tidal range reaching 4.2 m and encompasses a diverse range of habitat types including mangroves, fringing coral reefs and seagrass beds.

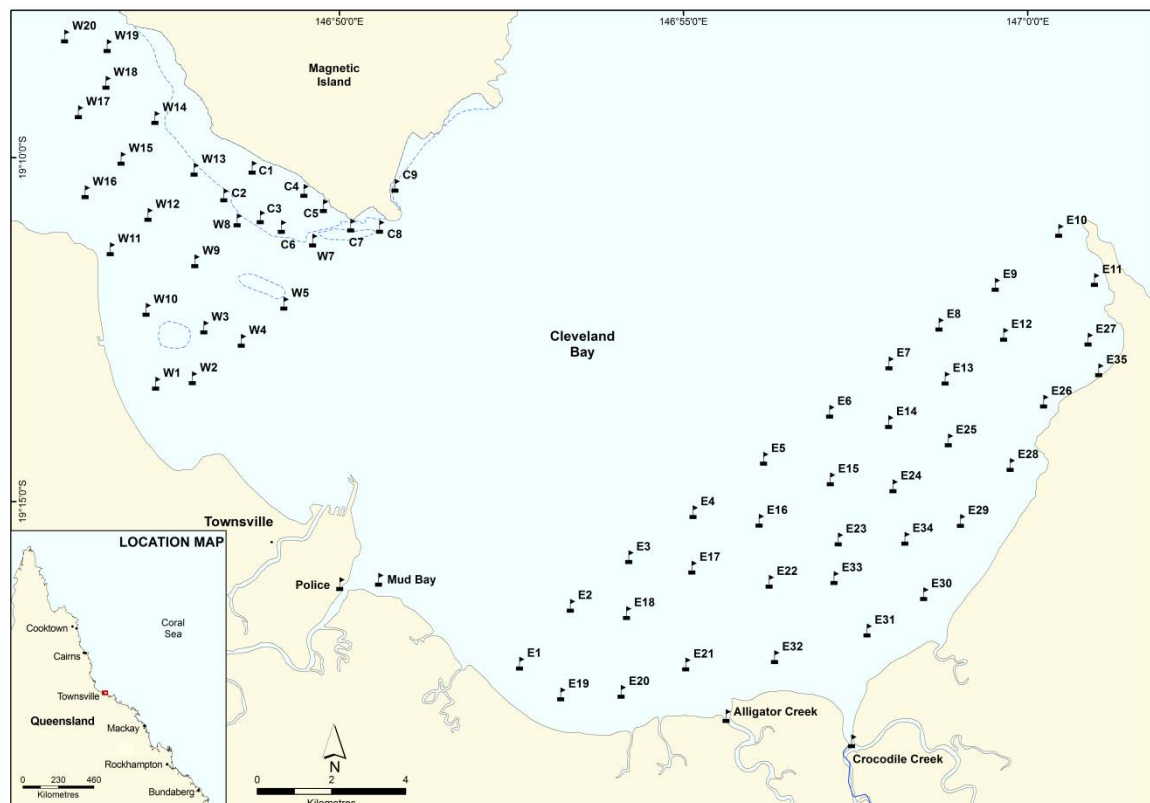


Fig 4.1 Cleveland Bay. Locations of acoustic receivers (W1-20) west side, (E1-34) east side and (C1-9) fringing reef, deployed in Cleveland Bay. Inset shows location of Cleveland Bay relative to the Queensland coast.

Field Methods

A series of 63 VR2W acoustic receivers (Vemco Ltd.; www.vemco.ca) were deployed in November 2008 throughout the Conservation Park Zone (gill-net and trawling prohibited, bait netting and line fishing permitted) of Cleveland Bay to passively monitor the movement of a range of inshore predators, including two shark-like batoids. Acoustic

receivers recorded time, date and identity of tagged individuals that swam within detection range of the units. Receivers were deployed in a grid arrangement and extended across all habitat types present including mangroves, seagrass, fringing reef, sand and mud. Receivers had a detection range of approximately 900 m (Heupel unpublished data). Receivers were serviced quarterly to download data, change batteries and remove biofouling.

Glaucostegus typus and *Rhynchobatus* spp. were caught between October 2009 and January 2011 using long-lines (500 m bottom set mainline – 6 mm nylon rope) and gill-nets (length 200 m, mesh size 114 mm). Hooks were attached to the long-line on gangions composed of a 1 m section of nylon cord, a swivel and 1 m of wire trace. Sizes 10/0, 14/0, 16/0 Mustard tuna circle hooks, 10/0 Gamakatsu octopus hooks, circle (Offset-Point) and 10/0 Eagle claw wide gap hooks were used and baited with squid (*Loligo opalescens*), blue threadfin salmon (*Eleutheronema tetradactylum*) or butterfly bream (*Nemipteris* spp.). Hook size and type was varied to reduce any size selectivity bias associated with the long-lines. Captured shark-like batoids were secured to the boat using a tail rope and then placed ventral side up. Once individuals were in a state of tonic immobility measurements and transmitter deployment commenced. Individuals were sexed, stretch total length (STL) was measured to the nearest mm, a genetic sample was taken and individuals were tagged with a rototag in the first dorsal fin. Individual maturity was classified as either juvenile or adult according to known size at maturity estimates (Last and Stevens 2009). Transmitters were surgically implanted into the abdominal cavity to ensure long-term retention and mitigate biofouling (JCU animal ethics permit #A1566). Individuals with stretch total lengths less than 700 mm were fitted with V13 transmitters (13 x 36 mm) and larger individuals were fitted with larger V16 transmitters (16 x 68 mm). All transmitters were coded to allow individual identification and were set to pulse randomly once every 45 – 75 s at 69 kHz. Random repeat rates allowed multiple individuals to be monitored simultaneously without the signals continuously overlapping.

Data Analysis

Data collected from acoustic receivers were analyzed to examine presence, residency and movement patterns of shark-like batoids within Cleveland Bay. The locations of monitored individuals within the receiver array were estimated every 30 min using a mean position algorithm that provided an individual's center of activity (COA) (Simpfendorfer et al. 2002a). Data analyses for this study were conducted in the R environment (R Development Core Team 2009).

Residency

The daily presence of shark-like batoids was defined by at least two detections of an individual for that day on any receiver within the array. Daily presence was plotted to provide a visually interpretable timeline of occurrence within Cleveland Bay throughout the study period. One-way ANOVAs were used to compare the influence of sex on total days monitored and total days detected. Total days monitored was defined as the total number of days from the first to last detection. A Residency Index (RI) was calculated for each individual following methods described by Simpfendorfer et al. (2011b) where the ratio between the number of days an animal was detected to the number of days from the first to the last detection was determined. A value of one indicated an individual was always present, while zero indicated an individual was not detected after release. Residency index values were compared between species with size, sex and total number of days monitored using analysis of covariance (ANCOVA). A post-hoc Tukeys unequal N Honest Significant Difference (HSD) test was used to identify groups that were significantly different from each other.

Activity Space

Activity spaces of shark-like batoids were calculated based on COA estimates using 50 and 95% kernel utilization distributions (KUD) calculated with the *adehabitat* package in R (Calenge 2006). Activity spaces were calculated at monthly intervals, plotted in R using Maptools and subsequently plotted using ARCMAP. One-way ANOVA was used to test for differences in 50 and 95 % KUDs within and between species, sexes and size classes. Monthly, size of KUD was compared between years for individual's for which philopatry was observed using one-way ANOVA.

4.3 Results

A total of 16 *G. typus* were fitted with acoustic transmitters and included 7 males and 9 females representing comparable length ranges (Table 4.1). With the exception of one female (Transmitter 56316; STL = 2660 mm), all tagged and released *G. typus* provided detection data. Twenty *Rhynchobatus* spp. were fitted with acoustic transmitters. Females dominated this sample (n = 18) and ranged from 860 to 2650 mm STL. Only two males (975 and 1500 mm STL) were captured, fitted with transmitters and released. One female *Rhynchobatus* spp. (Transmitter 56319; STL = 2260 mm) released with a transmitter 1.3 km from the outer line of the eastern side of the array and one female (Transmitter 56312; STL = 1420 mm) released close to the western boundary were never detected.

Table 4.1 *Glaucostegus typus* (GSR) and *Rhynchobatus* spp. (WSG) acoustically monitored in Cleveland Bay. Date of capture, side of bay captured, sex, stretch total length and total days detected are indicated.

Species	Transmitter number	Date tagged	Side of capture and release	Sex	Stretch total length (mm)	Total days present
GSR	56311	16/10/2009	West	F	2670	52
GSR	56316	27/10/2009	East	F	2660	0
GSR	56317	27/10/2009	East	F	2590	42
GSR	56314	3/11/2009	East	F	2110	70
GSR	56536	3/11/2009	East	F	2650	149
GSR	59608	21/05/2010	East	M	1040	198
GSR	59615	6/09/2010	East	M	698	20
GSR	59612	13/09/2010	East	M	2650	26
GSR	59610	15/09/2010	East	M	2450	21
GSR	59613	15/09/2010	East	M	2630	35
GSR	56544	28/10/2010	East	M	1450	401
GSR	63540	9/11/2010	East	F	680	2
GSR	56543	17/11/2010	East	F	2560	43
GSR	63543	17/11/2010	East	F	508	41
GSR	63541	17/11/2010	East	M	510	2
GSR	56539	7/12/2010	East	F	2650	5
WSG	56310	2/10/2009	East	F	1580	182
WSG	56318	27/10/2009	East	F	2250	81
WSG	56319	27/10/2009	East	F	2260	0
WSG	56538	3/11/2009	East	F	2210	90
WSG	56313	4/11/2009	East	F	2100	9
WSG	56533	5/11/2009	East	F	1780	4
WSG	56534	5/11/2009	East	F	2220	13
WSG	56315	25/05/2010	West	F	NA	100
WSG	56535	27/10/2010	West	F	860	1
WSG	56537	11/11/2010	West	F	2120	350
WSG	56312	1/12/2010	West	F	1420	0
WSG	56541	1/12/2010	West	F	1710	4
WSG	56540	7/12/2010	West	F	2050	2
WSG	59609	7/12/2010	East	F	2000	5
WSG	46976	13/12/2010	West	M	975	20
WSG	46977	16/12/2010	West	F	1540	13
WSG	46986	10/01/2011	West	F	1960	110
WSG	46974	18/01/2011	West	F	1530	252
WSG	46981	18/01/2011	West	F	1480	296
WSG	46978	18/01/2011	West	M	1500	182

Residency

Glaucostegus typus were monitored between 1 and 766 days (mean = 333 ± 69 days) and were present in the site from 2 to 401 days (mean 73 ± 25 days). There was no significant difference between sexes for either total days monitored (Table 4.2: ANOVA, $F_{1,13} = 0.66$, $P = 0.42$) or days present (Table 4.2; ANOVA, $F_{1,13} = 0.8237$, $P = 0.38$). The RI did not differ significantly between sexes (Table 4.2; ANOVA, $F_{1,13} = 0.23$, $P = 0.63$). Residency of both sexes changed with individual size (ANOVA, $F_{1,13} = 8.86$, $P < 0.05$). Juveniles (STL < 1000 mm) had very low residency indexes. Residency increased in sub adult individuals (STL 1000-1500) and then decreased for adults (STL > 1500mm).

Philopatric behaviour (returning to the same location in subsequent years) was exhibited by adult *G. typus*. Adult females were observed to leave the bay in the first weeks of December at the onset of the wet-season and returned in October the next year. Six of nine *G. typus* females ranging in size from 2110 to 2670 mm exhibited philopatry (Fig 4.2a). Periods of absence ranged from 284 to 704 days (mean = 391 days). Two of these individuals returned in two consecutive years; 56311 was absent between 5/11/2009 and 7/11/2010 (367 days) and again between 13/12/2010 and 28/10/2011 (319 days). Similarly, 56536 was absent between 14/12/2009 and 4/10/2010 (294 days) and again from 21/12/2010 to 2/10/2011 (285 days). The remaining four females had a single philopatric event during the monitoring period with absences of 284, 309, 383 and 704 days respectively. Three male *G. typus* were also observed to leave and return to the bay with periods of absence of 155, 286 and 333 days (Fig 4.2a). Males returned to the bay earlier than females, typical during August and September.

Table 4.2 Presence of *Glaucostegus typus* and *Rhynchobatus* spp. in Cleveland Bay, including sample size, number of days individuals were monitored and number of days detected, residency index, 50 % and 95 % KUDS

Species	Sex	n	Total days monitored			Residency Index			Center of activity					
			Min	Max	Mean	Min	Max	Mean	50% KUD (km ²)			95% KUD (km ²)		
			Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean
<i>G. typus</i>	Female	9	1	766	349	0	1	0.4	2.5	18.8	9.1	6.3	60.1	41.4
	Male	7	11	395	281	0.1	1	0.4	2.4	12.2	8.4	10.	63.9	41.9
<i>Rhynchobatus</i> spp.	Female	18	1	707	222	0	1	0.5	4.3	20.6	7.4	18.	76.4	33.7
	Male	2	20	208	114	0.3	0.5	0.4	5.3	7.03	4.8	18.	25.3	22.1

Rhynchobatus spp. were monitored for 1 to 707 days (mean = 231 ± 50 days) and were present in the site from 1 to 350 days (mean 82 ± 24 days) (Table 4.2: Fig 4.2b). There was no significant difference between sexes in total days monitored (Table 2: ANOVA, $F_{1,15} = 0.01$, $P = 0.91$) and total days present (Table 4.2: ANOVA, $F_{1,15} = 0.12$, $P = 0.72$). However, given the low number of males monitored these results are inconclusive. There was no significant difference in RI between sexes (Table 4.2: ANOVA, $F_{1,15} = 0.06$, $P = 0.80$) or size of individuals (ANCOVA $F_{3,12} = 0.8$, $P = 0.51$). Individuals were observed to leave Cleveland Bay and return again with absences ranging from days to months. The longest absence was by a female (Transmitter 46986; STL = 1960 mm) between 20/4/2011 and 13/11/2011 (207 days). However, synchronous philopatric behaviour was not evident for *Rhynchobatus* spp. individuals. Individuals of all sizes monitored intermittently left the array for short periods (days-weeks) prior to returning. With the

exception of individual 46986 no individuals were observed to return to the bay once they had been absent for more than 200 days.

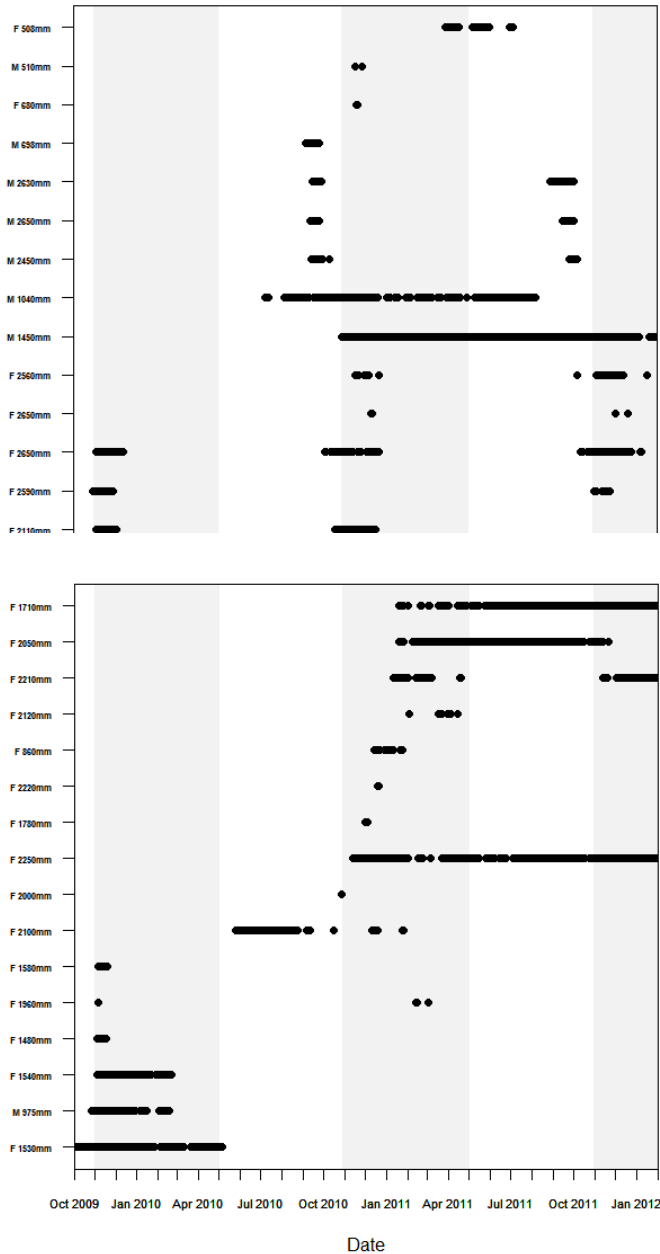


Fig 4.2 Presence of (a) *Glaucostegus typus* and (b) *Rhynchobatus* spp. by day in the study site of Cleveland Bay. Grey shaded areas represent the wet season which occurs between November and May. (M) males and (F) females, STL (mm).

Activity Space

When detected within the array *Glaucostegus typus* predominantly remained within the area of capture and displayed small core activity spaces (Fig 4.3). Only two individuals moved between the eastern and western side of the array, both were adults: one female (STL = 2670 mm) and one male (STL = 2450 mm). *Glaucostegus typus* were found to have monthly 50 % KUDs that ranged from 2.4 to 18.2 km² (mean = 9.57 km²) and monthly 95 % KUDs that ranged from 6.3 to 63.9 km² (mean = 43.38 km²). Females and males had similarly sized activity spaces (Table 4.3; ANOVA, 50 %: $F_{1,13} = 0.46$, $P = 0.5$; ANOVA, 95 %: $F_{1,13} = 0.00$, $P = 0.95$). *Glaucostegus typus* with lower residency indices had larger activity spaces (Fig 4.4a,b; ANCOVA 50 %: $F_{1,13} = 8.43$, $P < 0.05$; 95%: $F_{1,13} = 14.95$, $P < 0.001$) and activity space varied with the size of individual (Fig. 4.4 c,d; 50 %: $F_{1,13} = 9.11$, $P < 0.05$; 95 %: $F_{1,13} = 19.14$, $P < 0.001$). Juveniles (STL < 1500 mm) had activity spaces that were concentrated in the shallow regions of Cleveland Bay while adults used shallow areas in addition to deeper regions further from the coast. *Glaucostegus typus* returning to the bay annually used the same areas where they had been detected in previous years (Fig 4.5) and activity space was similar among years (Table 4.3; ANOVA. 50 %: $F_{3,3} = 2.178$, $P = 0.2696$; 95 %: $F_{3,3} = 4.42$, $P = 0.12$).

Table 4.3 Comparison of activity space size of 50 % and 95 % yearly KUDs for *Glaucostegus typus* (GSR) individuals that displayed philopatry.

Sex	Size (mm)	50% KUD			95% KUD		
		2009	2010	2011	2009	2010	2011
Female	2670	6.0	2.6	6	24.4	15.1	45.3
Female	2590	11.2		9.8	44.9		50.9
Female	2560		10.0	11.2		40.8	46.4
Female	2650	11.3	15.1		60.4	60.2	
Male	2450		5.9	14.9		29.5	56.9
Male	2650		12.3	17.1		60.6	70.0
Male	2630		8.3	13.7		52.0	64.1

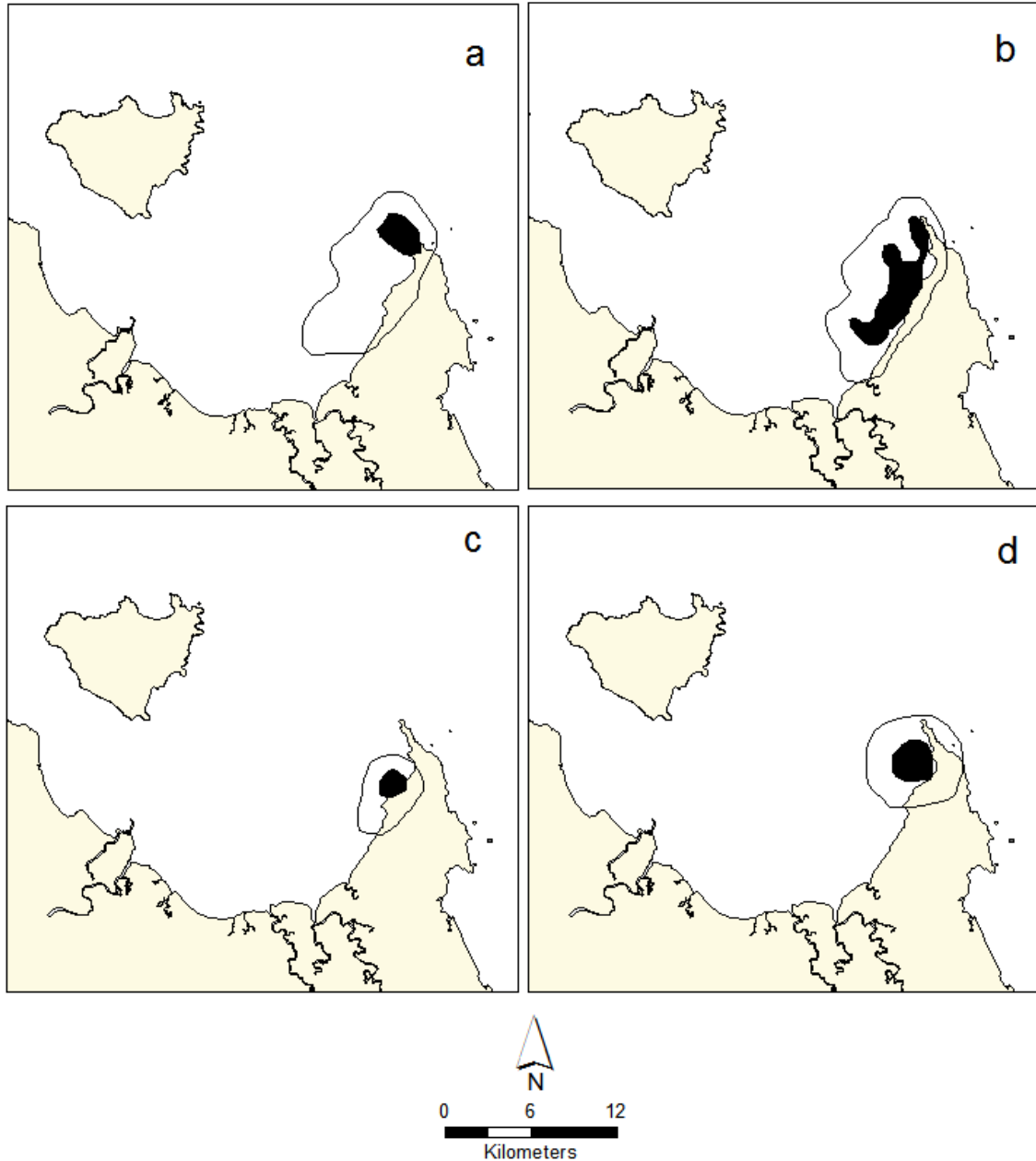


Fig 4.3 *Glaucostegus typus*. Yearly activity spaces of 4 *G. typus*, including (a) adult male (STL = 2630 mm), (b) adult female (STL = 2650 mm) (c) sub adult male (STL = 1450 mm), and (d) juvenile female (STL = 508 mm). Panels are 95 % Kernel Utilization distributions (KUDs) (solid line) and 50 % KUDs (black fill).

Rhynchobatus spp. activity space within Cleveland Bay tended to be localized within the western side of the bay (Fig 4.6). *Rhynchobatus* spp. 50 % KUDs ranged from 4.0 to 20.6 km² (mean = 7.03 km²) and 95 % KUDs ranged from 18.6 to 76.4 km² (mean = 41.04 km²). There was no significant difference in KUD size between sexes (Table 4.3; ANOVA, 50 %: $F_{1,15} = 0.45$, $P = 0.5$; ANOVA, 95%: $F_{1,15} = 0.82$, $P = 0.38$). Activity space size of *Rhynchobatus* spp. was not related to either RI (Fig. 4.4a,b. 50 %: ANOVA $F_{1,15} = 0.62$, $P = 0.44$; 95 %: $F_{1,15} = 0.36$, $P = 0.55$) or size of individual (4.4c,d. 50 %: ANCOVA $F_{1,14} =$

0.24, $P = 0.62$; 95 %: $F_{1,14} = 0.82$, $P = 0.37$). Although *Rhynchobatus* spp. preferred different regions of Cleveland Bay than *G. typus*, activity space size was similar (Fig 4.2; ANOVA, 50 %: $F_{1,32} = 1.31$, $P = 0.26$; ANOVA, 95%: $F_{1,32} = 2.03$, $P = 0.16$).

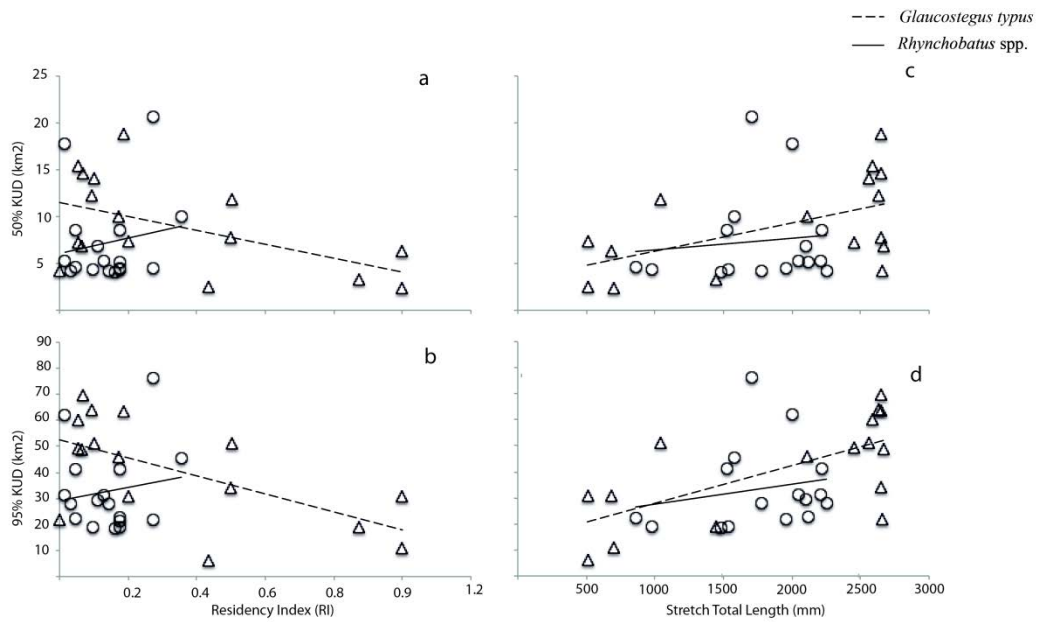


Fig 4.4 Relationship between residency index (RI) and activity space (KUD) size a) 50 % KUD, b) 95 % KUD. Relationship between individual size (STL) and activity space (KUD) c) 50 % KUD, d) 95 % KUD. Δ *Glaucostegus typus* (GSR), \circ *Rhynchobatus* spp. (WSG).



Fig 4.5 *Glaucostegus typus*. Yearly activity space of 4 adult *G. typus* that returned to Cleveland Bay inter-annually including; (a-c) female (STL = 2670 mm), (d-f) female (STL = 2650 mm), (g-h) male (STL = 2450 mm), and (i-j) male (STL = 2630 mm). Panels are 95 % Kernel Utilization distributions (KUDs)(solid line) and 50 % KUDs (black fill).

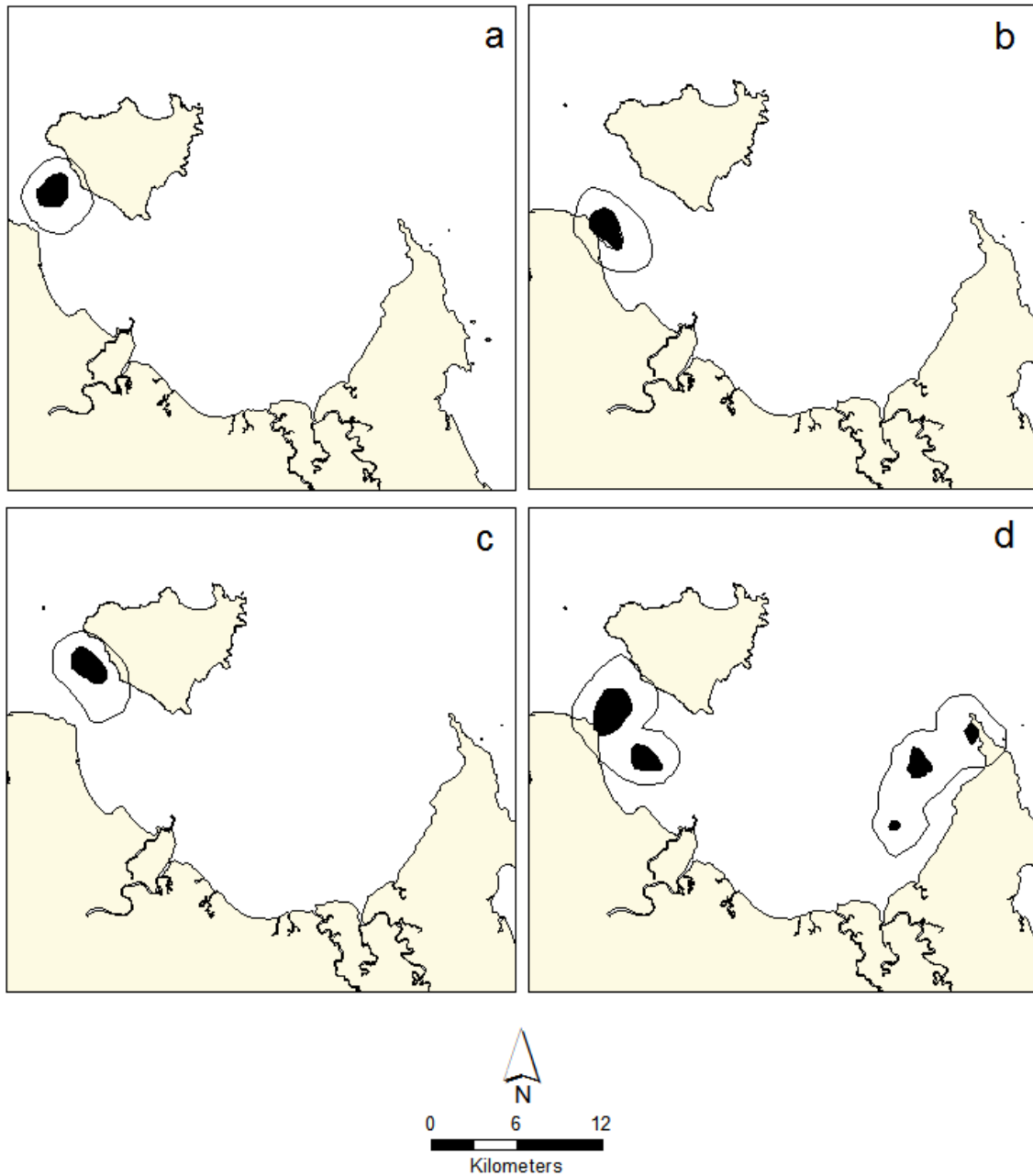


Fig 4.6 *Rhynchobatus* spp. Yearly activity space of 4 *Rhynchobatus* spp., including (a) male (STL = 975 mm), (b) adult female (STL = 1750 mm), (c) adult female (STL = 1960 mm), and (d) adult female (STL = 1710 mm). Panels are 95 % Kernel Utilization distributions (KUDs) (solid line) and 50 % KUDs (black fill).

4.4 Discussion

Using long-term movement data, this study found that despite being morphologically similar *Glaucostegus typus* and *Rhynchobatus* spp. use space in nearshore waters differently. *Rhynchobatus* spp. tended to be present for longer continuous periods while *G. typus* were present for shorter, predictable periods. Philopatry has been observed in a number of elasmobranch species (see review by Hueter et al. 2005) but the present study is the first to quantify the repetitive seasonal use of nearshore areas by a shark-like batoid. Individuals returned annually to use the same regions suggesting strong site fidelity. The spatial ecology of male *G. typus* changed with the ontogeny shift to maturity, with resident sub adult individuals (STL = 1000 – 1500 mm) becoming transient adults (STL = 2450 – 2650 mm). Reduced transmitter detection associated with very shallow habitats may have contributed to the low residency index values for individuals with stretch total lengths less than 1000 mm. Fisheries dependent (e.g. catch data; *R. productus*; Marquez-Farias 2005) and independent (e.g. belt transects; *G. typus*; Vaudo and Heithaus 2009) surveys have previously documented seasonal movement of shovelnose ray species into nearshore areas. However these studies did not quantify how individuals used space and the synchronous manner of the philopatry. *Rhynchobatus* spp. residency was highly variable with no relationship between individual size and presence within the bay. It is possible that any patterns of spatial ecology have been masked by monitoring individuals from all three species of the complex.

Glaucostegus typus showed both seasonality and site fidelity in the use of Cleveland Bay. Adult females arrived in October and left in the first weeks of December during the onset of the wet-season. Returning females inhabited the same regions of the bay and had similar sized activity spaces between years. Adult males returned to the bay several weeks prior to the return of females. Activity space of adult males and females overlapped during periods of presence within the bay. Fisheries independent sampling found adult males had sperm running and females of lengths over 2200 mm had mid- to late-term embryos between September to November (White unpublished data), suggesting that presence of adult *G. typus* within the bay may have been associated with pupping and possibly mating. Observation of neonates within mangrove habitats of the bay, after the wet-season further supports the link between use of the bay as a mating and/or pupping area. Other species of shovelnose ray (e.g. *Rhinobatos productus*) have been found in nearshore areas of California (Talent 1985) and Baja California (Salazar-Hermoso and Villavicencio-Garayzar 1999) during summer months suggesting that these species may seasonally migrate into these habitats. However, the longevity and intensity of the shovelnose ray fishery that operates in the area suggests the Baja California population

are resident year round and not philopatric (Farrugia et al. 2011). Traditional mark-recapture and acoustic monitoring of juvenile *R. productus* found no inter-annual site fidelity (Farrugia et al. 2011), similar to the present study in which only adults were observed to return to the study site. Strong site fidelity observed in adult *G. typus* suggests that nearshore areas are a key component of the species' spatial ecology, and may form critical habitat. Identification of critical habitats can greatly improve process of species management, through the use of spatial and seasonal regulations to protect both the habitats themselves and the species that use them.

The core activity space of *G. typus* juveniles was typically centred in shallow regions on the eastern side of the bay close to sand beaches and mangrove fringed coastline. Acoustic tracking of *Pristis pectinata* and *P. microdon* revealed similar behaviour with neonate sawfish inhabiting extremely shallow waters (Whitty et al. 2009, Simpfendorfer et al. 2010). The occurrence of *G. typus* in shallow waters may be related to predator avoidance, optimising growth or as a consequence of foraging behaviour (Sims 2003, Matern et al. 2004, Wethersee et al. 2007). Vaudo and Heithaus (2009) suggested that *G. typus* preference for shallow habitats in Shark Bay, Western Australia, was driven by physiological gains attained through the exploitation of local thermal heterogeneity. Physiological gains may also be driving habitat use of *G. typus* in the present study, but this remains to be demonstrated. While shallow nearshore habitats may provide advantages for shark-like batoids, their proximity to shore (and hence human development and activities) also makes them more vulnerable to anthropogenic impacts, and may mean that the species is most vulnerable in these habitats.

Glaucostegus typus and *Rhynchobatus* spp. are more mobile with larger activity spaces than other predominantly sedentary batoid species. With well-developed dorsal and caudal fins the body form of shark-like batoids falls between that of disc-shaped batoids and fusiform shark species. This morphology allows for greater swimming ability which likely contributes to larger activity spaces than reported for disc-shaped batoids. The activity space of benthically associated disc-shaped rays (e.g. *Dasyatis lata*, *Urobatis halleri* Cartamil et al. 2003, Vaudo and Lowe 2006), tend to be small (c. 1 km²), a consequence of spending long periods of time resting on the bottom. The fusiform shark species *Carcharhinus amboinensis* and *C. sorrah* monitored in Cleveland Bay had larger activity spaces (Knip et al. 2011a, Knip et al. 2012b) despite having smaller body sizes than the shark-like batoids examined. This suggests that shark-like batoids, while highly mobile, spend a portion of their time sedentary on the bottom and hence have moderate sized activity spaces. *Glaucostegus typus* and *Rhynchobatus* spp. spatial ecology is closest to morphologically similar species like the sawfish *Pristis pectinata* which has

reported activity spaces (95 % KUD) between 4 km² and 104 km² (Simpfendorfer et al. 2010), and mid-water swimming batoid species like the myliobatid ray *R. bonasus* which reportedly has an activity space between 0.1 km² and 62km² (Collins et al. 2007). *Glaucostegus typus* and *Rhynchobatus* spp. have smaller activity space sizes than highly mobile shark species, but larger than disc-shaped rays suggesting their behaviour lies somewhere between these two groups.

The lack of correlation between *Rhynchobatus* spp. size and residency may be a result of the occurrence of three possible species in the species group. However, there were two clusters of individuals of similar size but differing residency within these data that may represent different species within the *Rhynchobatus* spp. complex. Varying size at maturity between species may explain differences in residency, with individuals with higher residency belonging to a species with larger size at maturity (possibly *R. laevis*) and so monitored individuals would therefore be sub-adult. Large individuals with low residency may be adult *R. australiae* or *R. palpebratus*. The sample population was strongly skewed toward females, suggesting that habitat use may be partitioned by sex. With no general pattern of movement into or out of the bay it appears there is no synchronised philopatry as was the case for *G. typus*. Similar to *Rhynchobatus* spp. the fusiform shark species *Rhizoprionodon terraenovae* exhibited no consistent pattern of habitat use, had low residency and individuals moved into and out a bay frequently (Carlson et al. 2008). Like *R. terraenovae*, *Rhynchobatus* spp. may not be philopatric to specific nearshore areas but rather move between them.

Conclusion

The discrete use of nearshore areas has predominantly been described for fusiform shark species. Despite the ecological significance of shark-like batoids as meso predators there is little understanding of how and why they utilise nearshore areas. The results of this study show that shark-like batoids with similar morphology have differing spatial ecologies. Inter-annual consistency in activity space size and location within the bay, coupled with the reproductive stage of individuals suggest Cleveland Bay provides critical habitat for *G. typus*. *Rhynchobatus* spp. had different habitat use and residency in the bay. Further research is needed to quantify the movements, habitat preferences and seasonality of shark-like batoids in other regions and habitats if the spatial ecology of these species is to be fully understood at the ecosystem scale. The present study improves our understanding of shark-like batoid spatial ecology in nearshore waters and may provide potentially useful information for the management of these populations.

CHAPTER 5

Age and growth parameters of Australian shark-like batoids



Plate 5: *Rhynchobatus palpebratus*. Photo by Tiffany Sih (Townsville, 2012).

5.1 Introduction

Accurate estimates of life history are central to understanding species biology and implementing effective management and conservation initiatives (Walker 1998). Age and growth data inform decision-support tools such as ecological risk assessments (Braccini et al. 2006), demographic models (Cailliet 1992), stock assessments (Walker 1992) and ecosystem models (Stevens et al. 2000b, Simpfendorfer et al. 2011a) and therefore play a key role in the setting of controls to regulate fishing mortality (Simpfendorfer et al. 2011a). The low reproductive rates of many elasmobranch species (Hoenig and Gruber 1990) mean they can withstand only modest levels of fishing mortality (Camhi et al. 1998, Musick 1999, Cortes 2000). Therefore, interaction with fisheries must be carefully managed and a sound understanding of species life history strategy is integral to successful management (Simpfendorfer et al. 2011a).

In Australian waters shark-like batoids, such as the giant shovelnose ray *Glaucostegus typus* and the whitespotted guitarfishes *Rhynchobatus* spp. are considered to be less affected by fisheries than in the South-East Asian extent of their distribution (White and McAuley 2003a, b). The fins of shark-like batoids, known as “white-fin”, are highly prized and among the most lucrative of elasmobranch products (White and McAuley 2003a). Fishing effort is particularly intense in South-East Asia (Bentley 1996a, Chen 1996) where reductions in population size have been inferred by declining catch rates in the gill-net fishery (White and McAuley 2003a, b). Both *G. typus* and *Rhynchobatus* spp. have been assessed as globally Vulnerable by the International Union for Conservation of Nature (IUCN) (White et al. 2006). *Glaucostegus typus* and *Rhynchobatus* spp. have previously been classified as being at high risk of depletion from fisheries operating in northern Australian waters as they are susceptible to a range of gears, and there is little information regarding their biology (Anon 2003, Peverell 2005, Salini et al. 2007). The morphology of *G. typus* and *Rhynchobatus* spp. is unlike the disc-shaped body plans of most batoids and is akin to sharks with elongate body, well-developed caudal and dorsal fins and head, but with gill and mouth morphology similar to that of skates and stingrays. The shark-like body shape, may have implications for susceptibility to gear types with the dorsal fins increasing entanglement in gears that use mesh trawl nets, gill-nets, trap and seine nets (White et al. 2013a). Additionally, the high value of their fins increases retention rates in fisheries where these species are caught as bycatch.

Success in the management of elasmobranch fisheries is bound to the quality of biological information for the target and non-target species caught (Bonfil 2005). The

present study seeks to provide biological information to better inform the management of *G. typus* and *Rhynchobatus* spp. This study investigated the utility of vertebral analysis for determining age and growth in *G. typus* and *Rhynchobatus* spp. The specific aims of the present study were to (1) investigate vertebral structure of *G. typus* and *Rhynchobatus* spp. to determine its suitability for use in estimation of age, and (2) determine age and growth parameters.

5.2 Methods

Study species

Glaucostegus typus is the largest species of Rhinobatidae found in Australian waters, known to reach stretch total length (L_{ST}) > 2700 mm (Whitely 1939). Born at between 380 – 400 mm L_{ST} *G. typus* reach sexual maturity between 1500 – 1800 mm L_{ST} (Last and Stevens 2009).

Previously, a single name - *Rhynchobatus djiddensis* - was applied to all whitespotted guitarfish worldwide. However, recent taxonomic examination of Australian specimens revealed *R. djiddensis* does not occur in Australian waters (Last and Stevens 2009). Instead a complex of three morphologically similar species has recently been reported: *R. australiae*, *R. laevis* and *R. palpebratus* (Last and Stevens 2009). There is limited biological information available for members of this complex. Preliminary information suggests that *R. palpebratus* is the smallest of the three species with maximum c. 1000 mm L_{ST} (Compagno and Last 2008). *Rhynchobatus australiae* is the second largest of the three species, attaining L_{ST} > 2800 mm. Male *R. australiae* with L_{ST} > 1300 mm were found to be mature (White and Dharmadi 2007), whilst pregnant females were between 2800-3000 mm L_{ST} , with litters ranging from seven to 19 (mean = 14) embryos (White and Dharmadi 2007). *Rhynchobatus laevis* is the largest of the three species with a reported L_{ST} > 2700 mm (Whitely 1939b). Size estimates for each species are likely to vary particularly L_{ST} , due to the limited sample sizes and taxonomic confusion amongst individuals from which they were estimated. Size-at-birth for all three species is thought to be between 460-500 mm L_{ST} (White and Dharmadi 2007, Compagno and Last 2008).

Species identification within the complex relies on small differences in the number and pattern of white and black spots around the spiracles and gills as well as variation of fin size ratios. During the present study, visual identification of species using colouration was inconclusive and genetic analysis was unable to differentiate between *R. palpebratus* and *R. laevis* (Will White, CSIRO Hobart, pers. comm.). With continued taxonomic uncertainty the three species were treated as a single complex and subsequently all biological data

was pooled. This is in line with current management strategies for these species within Queensland commercial fisheries. Given that there are reported differences in size between species (Last and Stevens 2009), and that they are likely to have different growth rates, it was assumed the study would reveal large variation in the size at a given age.

Sample Collection

Samples were collected between March 2007 and February 2012 from commercial gill-net fishers operating in the Queensland East Coast Inshore Finfish Fishery (ECIFF). The ECIFF is a multi-species fishery targeting shark (primarily *Carcharhinus tilstoni* and *Carcharhinus sorrah*), grey mackerel (*Scomberomorus semifasciatus*), barramundi (*Lates calcarifer*) and threadfin salmon (polynemids: *Eleurotheronema tetradactylum* and *Polydactylus macrochir*). The fishery operates within the Great Barrier Reef World Heritage Area (GBRWHA) (from 10.5°S to 26°S). Fishers utilize a range of mesh sizes from a minimum of 114 mm, to a maximum of 165 mm stretched mesh. Fishery independent sampling using long-lines was conducted to supplement sample collection, particularly for size classes not encountered in the fishery (e.g. *G. typus* LST > 1500 mm White et al. 2013a).

Glaucostegus typus and *Rhynchobatus* spp. caught during fisheries dependent sampling were identified, sexed and measured in the field. Stretch-total length (L_{ST}) was measured in millimetres following Compagno (1984) by placing the animal belly down and depressing the upper lobe of the caudal fin into line with the body axis. Individuals caught in fisheries independent sampling were processed in the laboratory.

Vertebral processing and analysis

A section of five vertebrae was removed from the anterior region of the vertebral column between the gills and the first dorsal fin, and stored frozen. Vertebral samples were later thawed, cleaned of excess tissue, separated into individual centra and soaked in a solution of 5 % sodium hypochlorite (bleach) for approximately 30 minutes to remove remaining tissue, then rinsed thoroughly under tap water, and placed in a drying oven at 60°C for 24 hours. One of the five centra prepared from each individual was randomly selected for ageing. A slow speed saw with diamond tip blade was used to take a 400-600 µm longitudinal section through the focus of the centrum (Goldman and Musick 2006). This longitudinal section was mounted onto a microscope slide using thermoplastic cement (Crystalbond™ 509-1, SPI Supplies, Pennsylvania, USA). Longitudinal sections of vertebral centra were observed under a dissecting microscope using transmitted light

and photographed using a digital camera mounted to the microscope eye-piece. Age was estimated by counting the pairs of opaque and translucent bands deposited on the corpus calcareum. The birth mark (age = 0) was identifiable by a change of angle on the corpus calcareum. Any banding occurring prior to the birth mark was not included in age estimates (Goldman and Musick 2006). Age was determined as total number of bands occurring across the section after the birth mark. Two readers independently aged all centra twice in blind, randomized trials. Following Hoenig *et al.* (1995) and Evans and Hoenig (1998) systematic differences between reader's age estimates were tested using Chi-square tests of symmetry.

Age Validation

To test the assumption that the formation of growth-band pairs in *Rhynchobatus* spp. vertebrae are deposited annually, a fishery-independent mark-recapture study was carried out. Fishery-independent sampling using experimental multi-hook long-lines was conducted between January 2008 and February 2012 in Cleveland Bay (19 12° S; 146 54° E) near Townsville in north Queensland. The length and sex of captured individuals were recorded prior to external tagging on the first dorsal fin using Rototags or Jumbotags (Dalton, Worldwide). *Rhynchobatus* spp. vertebrae were chemically marked by injecting the fluorescent dye calcein (C₃₀H₂₆N₂O₁₃). Calcein was chosen over other dyes such as Oxytetracycline because it forms a mark that can be seen under a standard stereo microscope without the need for ultraviolet light. Twenty nine *Rhynchobatus* spp. females (STL = 800 - 2240 mm) and one male (STL = 1300 mm) were injected with a solution of 12.5 mg ml⁻¹ of calcein intramuscularly behind the first dorsal fin (McAuley *et al.* 2006). Two of the thirty calcein-marked *Rhynchobatus* spp. were recaptured as determined by a calcein mark clearly visible in their vertebrae (Fig 5.1). Both individuals had lost their external tags and it was therefore not possible to determine the exact date they had been chemically marked. Time at liberty was estimated by identifying all individuals smaller than the recaptured individual and counting the number of bands between the calcein mark and edge of the vertebrae. No *G. typus* were chemically marked during the study.

Vertebral samples from recaptured specimens that had been chemically marked were processed as described above. The distance between the centrum edge and the calcein mark was measured with an optical micrometre and the number of full growth bands (translucent and opaque pairs) between the calcein mark and centrum edge counted. To determine the periodicity of growth-band formation the slope of the regression between the number of post-calcein bands and the time at liberty (in years) was calculated

(Simpfendorfer et al. 2002c).

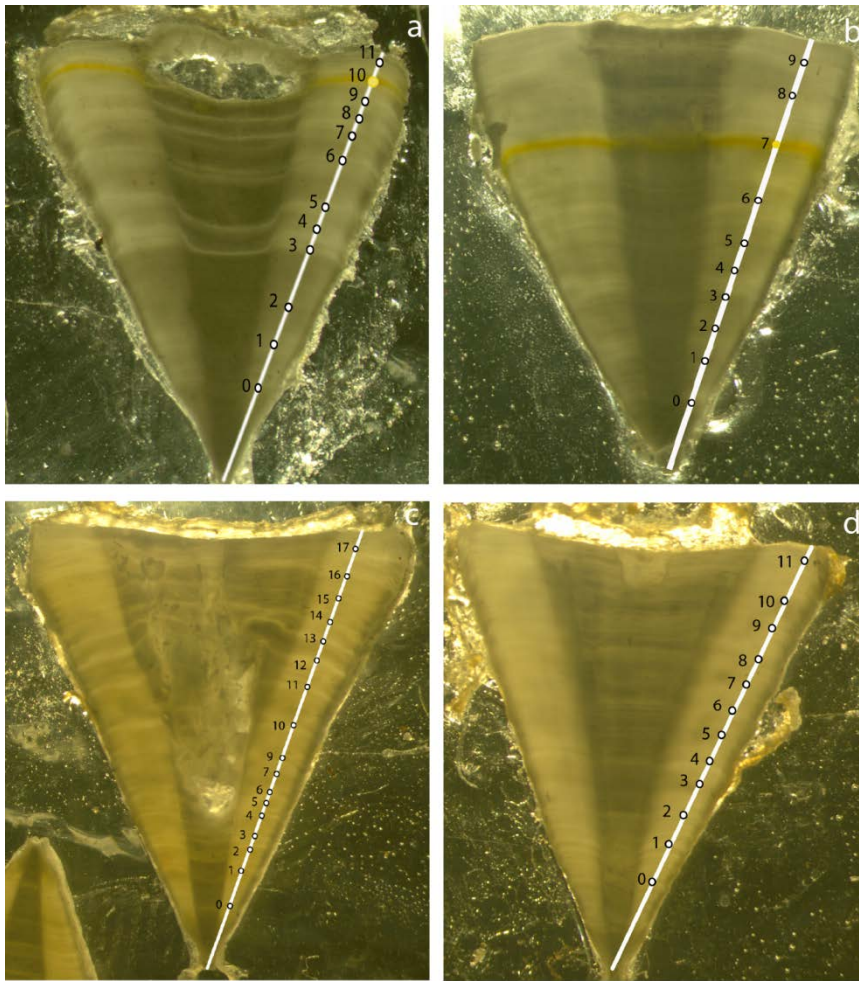


Fig 5.1 Sectioned vertebral centrum from two *Rhynchobatus* spp. (a) 1510 mm stretched total length female with eleven growth-band pairs visible, and (b) 2204 mm female with nine growth-band pairs visible. Translucent bands on the vertebrae are denoted by (●) and the calcein mark denoted by (●). Age 0 corresponds with the birth mark. Sectioned vertebral centrum from (c) 2840 mm stretched total length female *Glaucostegus typus* with 17 growth-band pairs visible and (d) 2320 mm male with 11 growth-band pairs visible.

Back-calculation

Back-calculation is used to describe the growth history of an individual (Goldman 2005), and has proven useful in understanding the life history of some species where sample sizes are limited or size classes are missing (Smart et al. 2013). Distances between growth band pairs were measured using a compound video microscope and an image analysis system, Image Pro Plus version 6.2 (Media Cybernetics 2002). The centrum radius (CR_c) was measured as the distance from the focus to the edge of the vertebra in a straight line. The distance from the focus to each of the growth band pairs (CR_i) and the

birth mark (CR_{birth}) were also measured along this line. All distances measured were to the nearest 0.001mm. Once measurements were taken the length-at-birth modified Fraser-Lee method (Campana 1990) was applied:

$$L_i = L_c + \left(\frac{(CR_i - CR_c) \times (L_c - L_{birth})}{(CR_c - CR_{birth})} \right)$$

where L_i was the estimated L_{ST} at band i , L_c the L_{ST} at capture, L_{birth} was the length-at-birth. This method was chosen as it incorporates exogenous information such as length-at-birth (Cailliet and Goldman 2004). The modified Fraser-Lee method is best used when the length-at-birth is known and does not match the intercept of a model applied with another method. Back-calculation was performed on 15 *Rhynchobatus* spp. vertebrae (950 – 2630 mm L_{ST}) and 16 *G. typus* vertebrae (636 – 2840 mm L_{ST}). Growth analysis used pooled data (sexes) and fixed values for size-at-birth: 500 mm L_{ST} (White and Dharmadi 2007) for *Rhynchobatus* spp. and 400 mm L_{ST} for *G. typus* (Whitely 1939b).

Model selection

A multi-model information theoretic approach to model selection incorporating Akaike's Information Criterion (AIC), was used to model the growth rates of *G. typus* and *Rhynchobatus* spp. The use of multiple models has been recommended over the use of a single model (Cailliet et al. 2006) and is proposed as an improvement over *a priori* use of the von Bertalanffy growth model (Katsanevakis and Maravelias 2008, Thorson and Simpfendorfer 2009). Six commonly used growth models in elasmobranch studies were used in the model selection process: 2-parameter von Bertalanffy (VBG2), 3-parameter von Bertalanffy (VBG3), 2-parameter Gompertz (GOM2), 3-parameter Gompertz (GOM3), a 2-parameter logistic model (LOG12), and a 3-parameter logistic model (LOG13) (see Thorson and Simpfendorfer 2009 for details of functions). All 2-parameter growth models had the size-at-birth as a fixed value. Candidate models were fitted to length-at-age data, with each model representing an alternative hypothesis for growth (Katsanevakis and Maravelias 2008).

Models were fitted using the method of non-linear least squares conducted in the R environment (R Development Core Team 2009). Given the small sample sizes an Akaike's Information Criterion with an incorporated bias correction algorithm (AIC_c) was used to evaluate model performance. This method has been shown to perform similar to Akaike's information criterion (AIC) when sample sizes are below 200 (Zhu et al. 2009). The AIC_c was calculated as:

$$AIC_c = AIC + \frac{2k(k+1)}{n-k-1}$$

where $AIC = n \log(\sigma^2) + 2k$, k is the total number of parameters and n is the sample size. The most appropriate model was the one with the lowest AIC_c value (AIC_{\min}). The AIC difference (Δ) was used to rank the remaining models and was calculated for each model ($i = 1-5$) as:

$$\Delta = AIC_c - AIC_{\min}$$

Models with the highest support had Δ of 0–2 while models with considerably less support had Δ of 2–10 and models with little or no support had $\Delta > 10$ (Burnham and Anderson 2002). AIC weights (w) were also calculated for each model ($i = 1-5$) and represent the probability of choosing the correct model from the set of candidates (Burnham and Anderson 2001, Braccini et al. 2007, Harry et al. 2011a).

This was calculated as:

$$w_i = \frac{\exp(-\frac{\Delta_i}{2})}{\sum_{j=1}^5 \exp(-\frac{\Delta_j}{2})}$$

A Kruskal-Wallis test was performed in the statistical package R to assess differences between groups for size-at-age for *Rhynchobatus* spp. A post-hoc test in the form of a multiple comparison tested if any of the sizes-at-age were significantly different from the rest. Ages 4, 8, 9, 10 and 11 were excluded due to limited sample sizes ($n < 3$).

5.3 Results

Vertebrae samples were obtained for 47 *Rhynchobatus* spp., 30 females (525–2630 mm L_{ST}), 12 males (520–1350 mm L_{ST}) and five individuals of unknown sex (1222–2130 mm L_{ST}) (Fig 5.2). The oldest male and female *Rhynchobatus* spp. aged were 5 years (1049 mm L_{ST}) and 12 years (1830 mm L_{ST}), respectively. The largest male and female *Rhynchobatus* spp. were 1350 mm L_{ST} and 2630 mm L_{ST} , respectively. Variation in length-at-age is likely indicative of the varying growth characteristics of the three species within this complex. The results of a Kruskal–Wallis test were significant ($H = 17.75$, $df = 2$, $P < 0.0001$) with *Rhynchobatus* spp. individual length-at-age significantly different between ages. Greatest variability was observed in length-at-age classes 2 (525 – 1130 mm L_{ST}), 3 (635 – 1610 mm L_{ST}) and 7 (1350 – 2140 mm L_{ST}) (0). The three year old age class had the greatest variability, but this may be the result of species specific differences in growth rates. Initial growth analysis using data pooled (species and sexes) and $L_{birth} = 500$ mm indicated the 2-parameter logistic (LOGI2) growth model had the highest AIC_w , however there was also support for the 2-parameter Gompertz (GOM2) (Table 5.1; Fig 5.4a). With $L_{birth} = 500$ mm the growth coefficient derived using von Bertalanffy for *Rhynchobatus* spp. was $k = 0.40 \text{ yr}^{-1}$. Growth analysis using back-calculated length-at-age data indicated that the 3-parameter Gompertz (GOM2) growth model had the highest AIC_w with both the 2-parameter Gompertz and 3-parameter logistic models also having strong support (Table 5.1; Fig 5.4b).

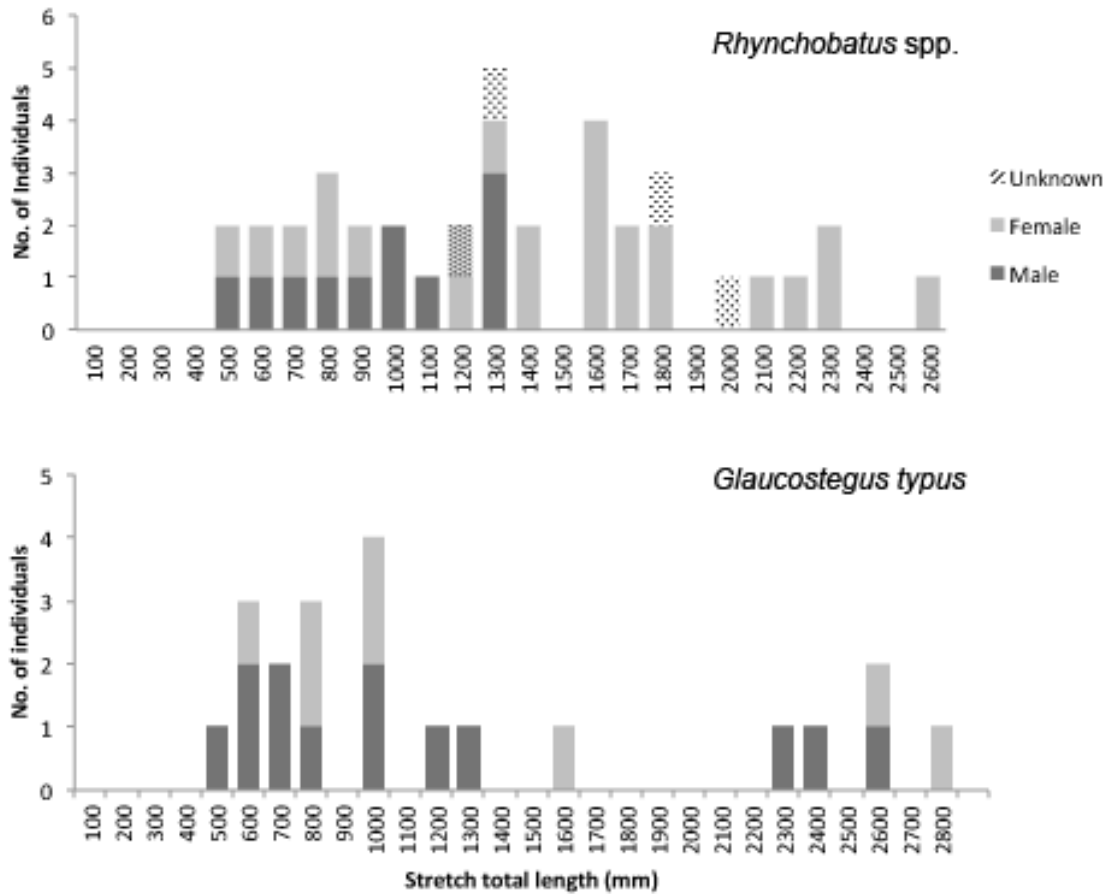


Fig 5.2 Stretched total length (L_{ST})-frequency distributions of (a) *Rhynchobatus* spp. ($n = 42$) and (b) *Glaucostegus typus* ($n = 23$) specimens collected off eastern Australia between March 2007 and February 2012

Table 5.1 Summary of six *a priori* growth models fitted to stretch total length-at-age data and back-calculated estimates (Back Cal.) for *Rhynchobatus* spp. and *Glaucostegus typus*. Models are ranked in terms of performance (best to worst with the best in bold) based on computed values of small-sample, bias-adjusted Akaike's information criteria (AIC_c). Akaike differences (Δ), Akaike weights (w) and residual standard error (RSE) show the relative support for models. Set of growth models used for multi-model inference, following Thorson and Simpfendorfer (2009) and Harry *et al.* (2011a).

Species	Model	Observed					Back-calculated					
		AIC _c	Δ	W	RSE	L _∞	AIC _c	Δ	W	RSE	L _∞	
<i>Rhynchobatus</i> spp.	VB2	703.9	1.7	13.8	355.2	2566.3	2572.1	8.00	0.58	240.9	2624.9	
		6	5	9	1	0	3		7	0		
	VB3	705.9	3.7	5.11	359.1	2571.3	2567.4	3.35	5.93	237.3	2293.6	
		5	5		4	2	8		5	4		
	GOM2	702.9	0.7	23.2	351.4	2179.7	2567.0	2.95	7.24	237.7	2071.5	
		3	2	4	2	0	8		2	1		
	GOM3	704.8	2.6	9.04	254.9	2235.4	2564.5	0.46	25.1	235.5	1960.8	
		2	1		0	2	9		4	1	9	
	LOGI	702.2	0.0	33.3	348.7	2045.4	2564.2	0.15	29.4	235.9	1896.8	
	2	0	0	8	8	7	8		3	4	7	
	LOGI3	703.7	1.5	15.3	351.0	2105.7	2564.1	0.00	31.6	235.2	1848.4	
		6	5	5	0	4	3		9	2	5	
<i>Glaucostegus</i> <i>typus</i>	VB2	982.7	0.6	21.5	323.0	3202.0	2273.0	1.39	26.5	152.5	2566.3	
		1	0	4	1	0	8		6	3	0	
	VB3	983.5	1.4	13.9	322.8	3808.0	2275.0	3.33	10.0	152.9	2571.2	
		8	8	2	0	0	2		7	4	4	
	GOM2	984.4	2.3	8.84	327.2	2597.1	2278.3	6.69	1.88	154.8	2179.7	
		9	9		7	0	7			4	4	
	GOM3	982.4	0.3	25.1	320.0	2919.6	2271.6	0.00	53.2	151.5	2235.4	
		0	9	6	0	1	9		6	0	0	
LOGI2	988.2	6.1	1.34	336.4	2371.4	2303.8	32.1	0.00	166.4	2045.5		
	7	6		9	0	7	8		7	0		
LOG1	988.1	0.0	29.2	319.3	2660.6	2275.4	3.73	8.23	153.1	2105.7		
	3	0	0	1	0	9	2		2	1		

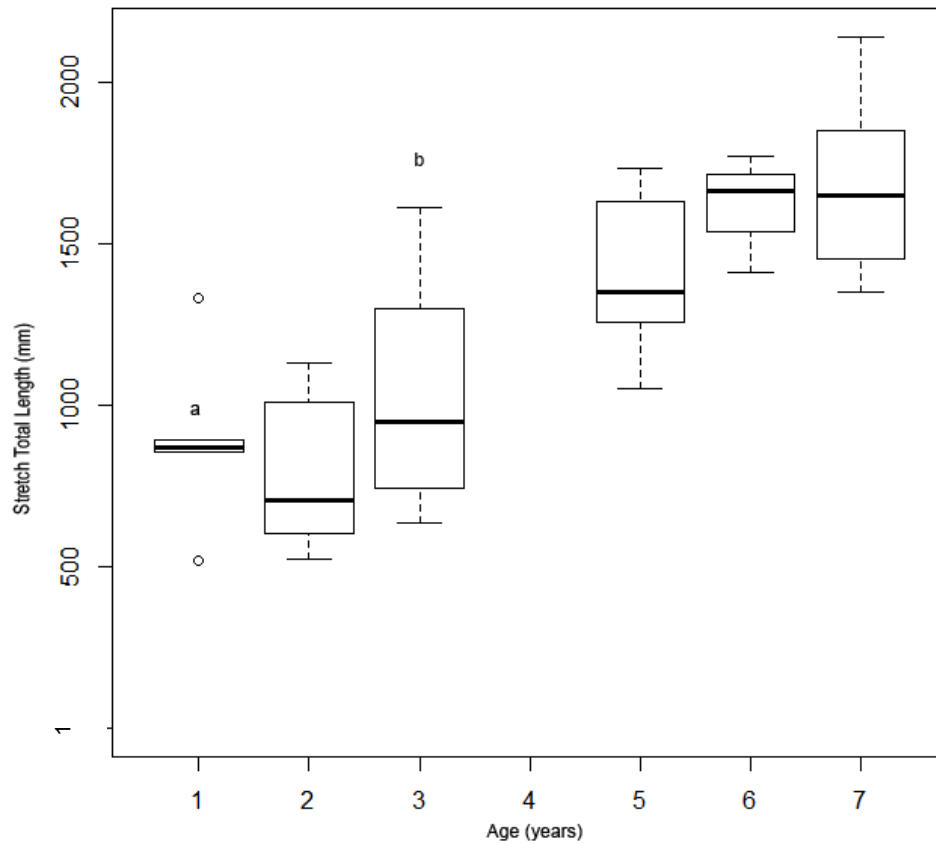


Fig 5.3 Box plot showing variation in size at age of *Rhynchobatus* spp., black line denote mean size at age, error bars indicate standard deviation. Size at birth for species within the *Rhynchobatus* spp. complex is similar (a), with variability peaking at age three (b).

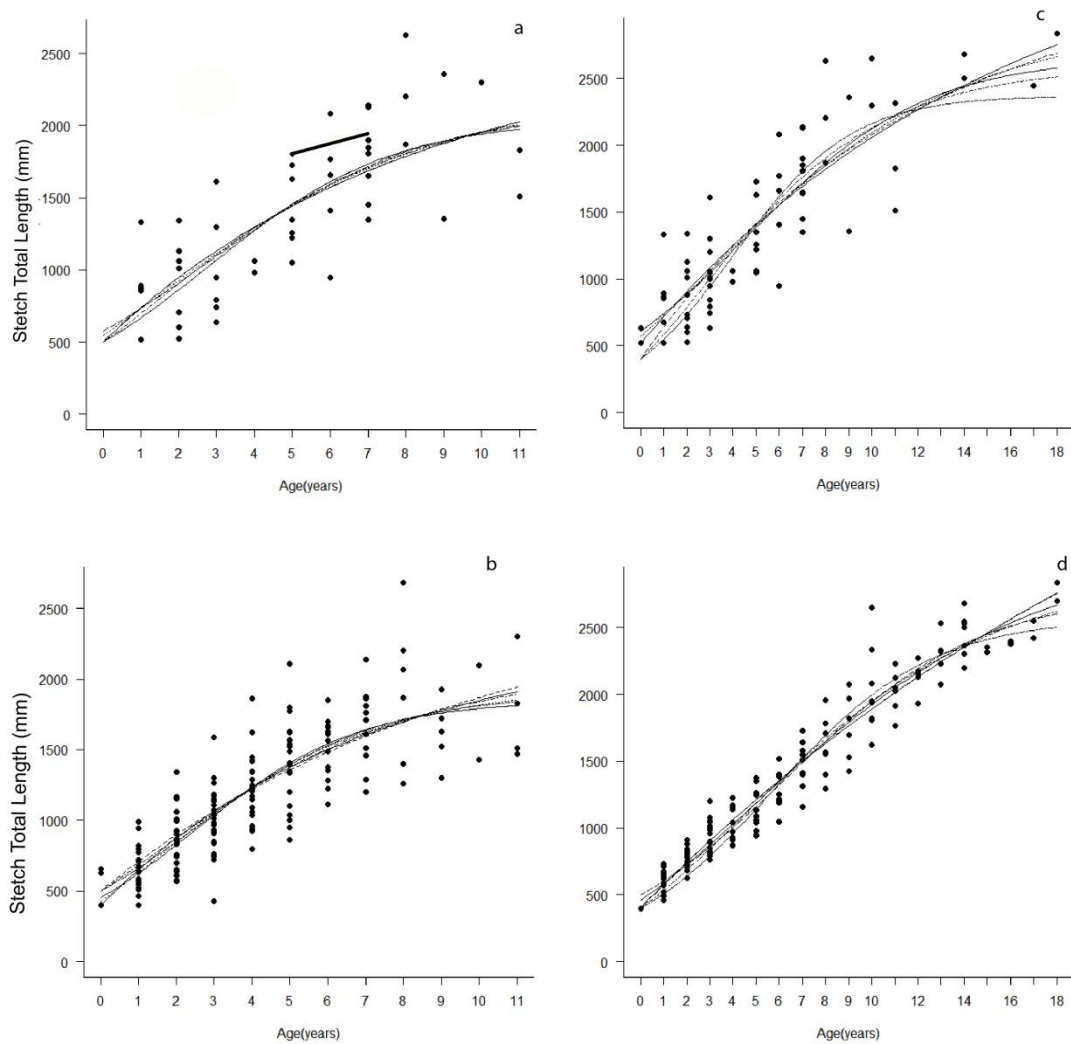


Fig 5.4 Fitted growth models for *Rhynchobatus* spp. (a,b) and *Glaucostegus typus* (c,d) using observed (a,c) and Fraser Lee length-at-birth modified back-calculated (b,d) data. Solid black line between age classes five and seven indicate growth of recaptured individual J0149

Vertebrae samples were obtained from 23 *Glaucostegus typus*, 9 females (640–2840 mm L_{ST}), 14 males (742–2650 mm L_{ST}) (0). The oldest male and female *G. typus* aged in the present study were 17 (2450 mm L_{ST}) and 19 years (2500 mm L_{ST}), respectively. The largest individual (2840 mm L_{ST}) was a female with an estimated age of 18 years. Initial growth analysis using pooled data (sexes) and $L_{birth} = 400$ mm (Whitely 1939b) suggested a 3-parameter logistic (LOGI3) growth model had the greatest support (Table 5.1; Fig 5.4c). Two growth models had AIC difference (Δ) < 2, while four models had > 10 % support (Table 5.1). At L_{birth} 400 mm the growth coefficient derived using von Bertalanffy for *G. typus* was $k = 0.15 \text{ yr}^{-1}$. Growth analysis using back-calculated length-at-age data with L_{birth} of 400 mm indicated that 3-parameter Gompertz (GOM3) was the preferred

model, however there was also support for the 2-parameter von Bertalanffy (VBG2) model (Table 5.1: Fig 5.4d).

The increase in sample size provided by back-calculation for both *Rhynchobatus* spp. and *G. typus* resulted in lower standard errors around the model parameter estimates (Table 5.1). However, despite greater statistical support, back-calculated models tended to underestimate L_{∞} for both *Rhynchobatus* spp. and *G. typus*, with the observed data set providing a more reasonably biological representation of L_{∞} despite the sample size. Back-calculated growth curves for *Rhynchobatus* spp. (0 a, b), appeared to show a two-phase growth pattern in addition to a greater variability in estimates of L_{∞} in comparison to models generated using observed data. In contrast, the growth curves produced for *G. typus* (0 c, d) showed little difference between models generated using observed and back-calculated data.

Opaque bands were apparent and easily discernible in vertebrae sections from both *Rhynchobatus* spp. and *Glaucostegus typus* (Fig 5.1). There was no significant difference in vertebral counts between readers ($\text{Chi}^2 X^2 = 17.58$, $\text{df} = 44$, $P = 0.99$). Calcein marking of *Rhynchobatus* spp. occurred between October 2008 and September 2009. Female 208391 (2204 mm L_{ST}) was recaptured October 2011 within Cleveland Bay where it was originally marked. Twenty individuals between 1520-2120 mm L_{ST} were chemically marked between April 2008 and August 2009; this gives possible time at liberty between 2.2 - 3.5 years (791 - 1308 days). Two bands were apparent between the edge and the calcein mark (Fig. 4). Female 356649 (1510 mm L_{ST}) was recaptured in July 2011, 160 km north of release location. Only six calcein injected animals had $L_{\text{ST}} < 1500$ mm at time of injection, these individuals were injected between September 2008 and September 2009 giving a possible time at liberty between 1.8 - 2.8 years (668- 1033 days). One band was evident after the calcein mark (Fig 5.1). Annual growth-band deposition thus could not be excluded in *Rhynchobatus* spp. using mark-recaptured individuals. Individual (J0149) female (1710 mm L_{ST}) was first captured 28/10/2008 and subsequently recaptured 10/1/2011, 804 days later having grown 100 mm (1810 mm L_{ST}), giving a growth rate of 45.4 mm/year, this animal was not chemically marked.

5.4 Discussion

The present study provides estimates of age and growth for commercially valuable and ecologically vulnerable *G. typus* and *Rhynchobatus* spp. Data pertaining to the biology of elasmobranch species tends to be biased toward shark species, with relatively few

studies examining the biology of Rhinobatidae and Rhynchobatidae species. Despite being morphologically similar, the biology of *G. typus* and *Rhynchobatus* spp. are quite different. This is consistent with variations in life history of other morphologically similar groups of elasmobranch (Harry et al. 2011a, Harry et al. 2012). *Glaucostegus typus* individuals in the sampled populations were larger and slower growing in comparison to *Rhynchobatus* spp.

Data deficiency, especially in terms of life history, has been a major impediment to the management of many elasmobranch species (Simpfendorfer et al. 2011a). Further, it is often the most threatened, naturally rare or over-exploited species for which information is lacking (Smart et al. 2013). The sample sizes in the present study were smaller than those that give robust estimates of age and growth parameters. Minimum sample sizes of 200 have been suggested for best results (Kritzer et al. 2001, Thorson and Simpfendorfer 2009). However, more recently Smart *et al.* (2013) demonstrated that small samples can be used to derive adequate growth parameters. Back-calculation was used to increase the number of length-at-age data points in this study, effectively increasing the sample size used to model growth (Cailliet and Goldman 2004). However, as was the case in Smart *et al.* (2013), models derived from back-calculated data did not always provide the most biologically appropriate parameters of growth. Back-calculation can contribute to the exploration of life histories, but its use must be assessed on a species by species basis. Given the greater variability in length-at-age for *Rhynchobatus* spp., modelling observed data may be preferential despite greater statistical support for back-calculated models until species-specific length-at-age estimates can be validated.

Sigmoid growth functions (Gompertz and logistic) best described the growth of *Rhynchobatus* spp. and *G. typus*, providing the best statistical fit for both. Small sample size, particularly of the smallest and/or largest individuals can result in overestimates of L_{∞} when using the von Bertalanffy models (Cailliet 1990, Francis and Francis 1992, Goldman 2005) and this may explain the limited support for von Bertalanffy growth functions in this study. Further, von Bertalanffy growth models assume a constant decrease in growth rate with increasing age (Ricker 1979, Araya and Cubillos 2006, Dale and Holland 2012), and therefore do not account for changes in growth rate associated with changing life history stages (e.g. energy allocation to reproductive development or gestation), or environmental conditions (changes in temperature, and prey availability) (Araya and Cubillos 2006, Braccini et al. 2007, Dale and Holland 2012). There is increasing support for the use of sigmoid growth functions in elasmobranchs (Dale and Holland 2012), particularly for batiod species such as *Rhinoptera bonasus*, *Pteroplatytrygon violacea* and *Raja binoculata*, which may increase in mass at a greater

rate than either width or length (Cailliet and Goldman 2004, Neer and Thompson 2005). Similarly, for species where growth may occur at different rates through life e.g. slow growth during early and late stages of life, with fast growth mid-life (Carlson and Baremore 2005, Braccini et al. 2007) sigmoid curves may be appropriate. However, dismissing von Bertalanffy growth functions or drawing conclusions regarding growth phases is premature given the small sample sizes used in this study and the relatively similar Δ values from the multi model approach. The findings for the *Rhynchobatus* spp. are further complicated by the pooling of three species. The published biological information suggested large size differences and distinct patterning between species (Whitely 1939, Compagno and Last 1999, 2008, Last and Stevens 2009). However, these were less obvious in the field. Currently, the complex is managed as a single species and as such having a general growth model may prove useful in management decision making. The limited sample sizes have provided a preliminary examination of age and growth; however, larger sample sizes will provide greater certainty and may lead to different conclusions about the best model to represent growth.

Until species-specific identification is possible, defining the life history of *Rhynchobatus* spp. will remain complicated. It is probable that the collected samples included representatives from each species. *Rhynchobatus australiae* and *R. laevis* are the largest of the three *Rhynchobatus* spp. species thought to inhabit Australian waters attaining estimated sizes of 1870 and 2700 mm L_{ST} respectively. Given that female *R. australiae* can attain L_{ST} 3000 mm in Indonesia (White and Dharmadi 2007), the samples collected in the present study may not provide a full representation of growth for this species in Australian waters. The uniform size of individuals aged 0-1 in the present study may be an artifact of small sample sizes; however, taxonomic and field examination of female reproductive development suggests there are similar sizes-at-birth between species within the complex (White and Dharmadi 2007, Compagno and Last 2008). The increasing variability of length with increasing age may reflect the variability in growth between the smallest species *R. palpebratus* and the larger-bodied *R. australiae* and *R. laevis*. Further work is required to resolve the identification within the complex and define species-specific life histories.

Glaucostegus typus is the largest species in the family Rhinobatidae, attain lengths exceeding 2700 mm L_{ST} (Whitely 1939b), with individuals exceeding 2800 mm L_{ST} in the present study. There are few assessments of rhinobatid species biology, with previous work examining smaller species (e.g. *Rhinobatos productus* Timmons and Bray 1997). Frisk et al. (2001) found that larger-bodied elasmobranchs $L_{ST} > 2000$ mm typically had von Bertalanffy growth rates (k) $< 0.1 \text{ year}^{-1}$ and were more vulnerable to exploitation.

Growth rates derived from von Bertalanffy growth functions for *G. typus* ($k = 0.15 \text{ year}^{-1}$) were higher than the trend suggested by Frisk *et al.* (2001), indicating this species may be more resilient to population decline under exploitation than other large elasmobranchs. Alternatively, differences between *G. typus* and those found by Frisk (2001) maybe the result of the low number of species for which information was available to model. Investigations of other batoid species using von Bertalanffy growth functions have found wide variability in growth rates from fast growing species of skates ($k = 0.1 \text{ year}^{-1}$) (Cailliet and Goldman 2004), and sawfish ($k = 0.14 \text{ year}^{-1}$) (Simpfendorfer *et al.* 2008) to slower growing species of stingray ($k = 0.05 \text{ year}^{-1}$) (Smith *et al.* 2007). Timmons and Bray (1997) found *R. productus* had a low growth coefficient ($k \text{ female} = 0.01 \text{ yr}^{-1}$; $k \text{ male} = 0.09 \text{ yr}^{-1}$) and longevity of 11 years. The L_{ST} (1500 mm) of *R. productus* is half that of *G. typus*; following the classification of Frisk *et al.* (2001) the smaller species would be expected to have the greater growth rate. Thus, *G. typus* may not follow the proposed Frisk model. This would not be uncommon though because extensive work with carcharhinid, sphyrnid and laminid species has shown wide variability in life history parameters of growth and longevity between species within families. Further work examining rhinobatid species using a range of growth curves is required to quantify species-specific variations in biology.

The results of this study suggest that although morphologically similar, *G. typus* and *Rhynchobatus* spp. have differing life histories. In north-eastern Australian waters, *G. typus* were larger, grew slower and had greater longevity than *Rhynchobatus* spp. Direct comparisons of *G. typus* with individual species within the *Rhynchobatus* spp. complex are problematic given taxonomic uncertainty of the group. A clearer definition of taxonomic boundaries between species in the complex is required to improve life history knowledge and thus management. This study furthers the understanding of shark-like batoid biology and provides estimates of essential biology required for quantitative assessment of fisheries vulnerability and ultimately population stability.

CHAPTER 6
Understanding the vulnerability of shark-like batoids using
demographic analysis



Plate 6. Releasing juvenile *Rhynchobatus laevis*. Photo by Fernanda Defaria (Cleveland Bay, 2011).

6.1 Introduction

Globally batoids have become an increasingly large component of fisheries catch and in some cases have developed into target species in fisheries where they were once considered by-product or bycatch (Dulvy et al. 2014). Most often batoids are taken in complex multi-species fisheries where regulating catch of individual species is difficult. Depletion, and in some instances collapse, of large-bodied batoid populations, including species of skates (Brander 1981, Walker and Hislop 1998a, Dulvy et al. 2000, Frisk et al. 2001) and sawfishes (Thorson 1982, Simpfendorfer 2000, Carlson et al. 2007), indicate that at least some batoid species are highly susceptible to overfishing (Frisk et al. 2002). Traditionally, batoids have been of low economic value, caught as bycatch species, and thus, there is limited historic data with which to assess population dynamics. In the absence of quantified life history characteristics for many batoids, the potential for population depletion is high. Although few batoid species are caught in directed fisheries, those that are have produced boom and bust fisheries with few if any sustainable (e.g. *Pristis perotteti* in Lake Nicaragua (Thorson 1982); *Rhynchobatus* spp. in the Aru Islands (Chen 1996)).

Shark-like batoids (families Rhinobatidae, Rhynchobatidae, Rhinidae, Pristidae) are taken in fisheries for their flesh and fins, and are recognised as some of the most threatened families of elasmobranches world-wide (Dulvy et al. 2014). Shark-like batoids are caught by a variety of fishing gears including trawl, gill-net, trap and seine nets and hooks (Chen 1996, White and McAuley 2003a, White et al. 2013a). Fins of shark-like batoids, known as “white-fin”, are highly prized in Asian markets and are among the most lucrative of elasmobranch products (White and McAuley 2003a, Clarke et al. 2006a, Clarke et al. 2006b, Compagno et al. 2006b) and the “white-fin” is the key driver in their retention by fishers. Fishing effort for shark-like batoids, particularly using gill-nets, is intense in South-East Asia (Bentley 1996b, Chen 1996) where reductions in populations have been inferred from declining catch rates (White and McAuley 2003a, b). For example, the gill-net fishery around the Aru Islands grew rapidly from its inception in the 1970s, reaching a boom of 500 boats in the 1980s. Catch rates have steadily declined since, as have the number of boats operating in the fishery (Chen 1996). There are no target fisheries for shark-like batoids in Australian waters, but rhinobatids, rhynchobatids and rhinids are taken as bycatch in trawl and gill-net fisheries (Stobutzki et al. 2002, Zhou and Griffiths 2008, Harry et al. 2011b) throughout their ranges. It is unlikely that these species would become targeted in Australian waters given current regulations, but given their susceptibility to multiple fishing gears and the value of their fins; it is likely that they will continue to be retained when caught. Australian populations may be a good source of

information to help improve management for stocks in areas where populations have been depleted as a consequence of fishing.

Four species of shovelnose ray (family Rhinobatidae) and wedgefish (family Rhynchobatidae) occur in northern Australia: *Glaucostegus typus*, *Rhynchobatus australiae*, *R. laevis* and *R. palpebratus*. Three of these species (*Glaucostegus typus*, *R. australiae* and *R. laevis*) have been assessed as globally Vulnerable by the International Union for Conservation of Nature (IUCN) (White et al. 2006), while the fourth species has not been assessed. *Glaucostegus typus* and *Rhynchobatus* spp. have previously been classified as being at high risk of depletion by fisheries operating in northern Australian waters as they are susceptible to a range of gears, and there is little information regarding their biology (Peverell 2005, Salini et al. 2007). However, large catches of these species have not been reported, and more detailed assessment has indicated that while these species are available to fisheries their mobility, habitat use and the configurations of fishing gears used may buffer against high mortality (Zhou and Griffiths 2008, White et al. 2013a).

Demographic analysis is a commonly used decision support tool that has played a key role in the setting of output controls (e.g. size limits) to regulate fishery mortality (Simpfendorfer et al. 2011a). With little resilience to fishing mortality (Hoenig and Gruber 1990), many elasmobranch species can withstand only modest levels of fishing pressure (Camhi et al. 1998, Musick 1999, Cortes 2000). Demographic techniques can provide information on population recovery times and define ontogenetic stages where conservation efforts will be most effective in developing management measures for exploited or threatened species (Simpfendorfer 2000).

Reductions in population size and distribution of *G. typus* and *Rhynchobatus* spp. throughout the South-East Asia have been the result of unregulated fishing and a poor understanding of population vulnerabilities. Given conservation concern, paucity of biological data and documented interaction with multiple Australian fisheries (Stobutzki et al. 2002, Zhou et al. 2009, White et al. 2013a), an exploration of population demography will enable suitable management options to be considered. The purpose of this study was to explore the demography of *G. typus* and *Rhynchobatus* spp. to define population parameters (e.g. rates of population increase, generation time), examine the effect of uncertainty in biological parameters on population parameters, and investigate the utility of sized-based management approaches.

6.2 Methods

Population dynamics of *G. typus* and *Rhynchobatus* spp. were investigated using age-based, matrix population models following Caswell (2001). Models were constructed using the PopTools add-in in MS Excel. Initially, age-specific life history information of *G. typus* and members of the *Rhynchobatus* complex (*R. australiae*, *R. laevis*, *R. Palpebratus*) was organized into the form of a life-table, with vectors of survival, and fecundity (Stearns 1992). No empirical natural mortality rate (M_x) estimates were available for either *G. typus* or *Rhynchobatus* spp. Values of M_x were instead estimated using Jensen's (1996) indirect age-independent method.

Matrix population model

Leslie matrices allow population dynamics parameters including mean generation length (G), net reproductive rate (R_0), intrinsic rate of growth (r) and instantaneous rate of population growth (λ) to be calculated for individual species (Krebs 2008). Life tables were re-arranged into a Leslie Matrix for statistical analysis (Caswell 2001) where the projection matrix, A is given by

$$A = \begin{array}{cccccc} & f_0 & f_1 & f_2 & f_3 & \dots & f_{x-1} \\ s_1 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & s_2 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & s_3 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & \dots & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & s_{x-1} & 0 & 0 \end{array}$$

where f and s are the age-specific fecundity and survival in a birth-pulse population with a pre-breeding census, respectively (Caswell 2001). The infinite rate of population growth, λ , is the dominant eigenvalue of A , where A is the a population projection matrix (Caswell 2001) and the population doubling time is calculated as

$$t_2 = \log_{\lambda} 2$$

*Life history data**Rhynchobatus* spp. life history

Previously, a single name: *Rhynchobatus djiddensis* was applied to all whitespotted guitarfish worldwide. However, recent taxonomic examination of Australian specimens revealed *R. djiddensis*, found in the Western Indian Ocean from the Red Sea to the Eastern Cape off southern Africa, does not occur in Australia (Last and Stevens 2009). In Australian waters, a complex of morphologically-similar species have been reported: *R. australiae*, *R. laevis* and *R. palpebratus* (Last and Stevens 2009).

There is limited published life history information available for *Rhynchobatus* spp. (Table 6.1). Published biological information suggested large size differences and distinct patterning between species (Whitely 1939, Compagno and Last 1999, 2008, Last and Stevens 2009), however, these have been less obvious in field observations (White et al. 2014). Preliminary age and growth data have been attained from the Queensland population, while estimates of size and reproductive maturity have been made from populations in Indonesia (White and Dharmadi 2007, White et al. 2014). *Rhynchobatus palpebratus* (maximum reported length <1500 mm L_{ST}) is the smallest of the three species. *Rhynchobatus laevis* is reported to be the largest species with a maximum size exceeding 2700 mm (Last and Stevens 2009) however, evidence from Indonesia indicates *R. australiae*, also attains sizes between 2700-3000 mm L_{ST} (White and Dharmadi 2007). The only age estimates of *Rhynchobatus* spp., come from the Queensland population where the maximum age of males and females were found to be 5 (1049 mm L_{ST}) and 12 years old (1830 mm L_{ST}), respectively (Table 6.1) (White et al. 2014). The largest male and female *Rhynchobatus* spp. sampled in the Queensland population were 1350 mm L_{ST} and 2630 mm L_{ST} , respectively (White et al. 2014). Preliminary assessments of the complex suggests considerable variability in size at age among species, in addition to size variability between sexes, thus size alone is not sufficient at differentiating between complex members (White et al. 2014). Currently, the complex is managed as a single species in Australian waters. White and others (2014) suggested that a general growth model may prove useful in management decision making in the absence of species-specific life history information, and in a fishery where all complex members are caught, sometimes with more than one species in a single net (White et al. 2013a).

Table 6.1 Biological information used to create Leslie matrices for *Glaucostegus typus* and *Rhynchobatus* spp. Fecundity and breeding frequency for *G. typus* are based on known values for *Rhinobatos productus* (Villavicencio 1993 , Márquez-Farías 2007). Reproductive information for *Rhynchobatus* spp. taken from White and Dharmadi 2007. Maximum age values taken for both *G. typus* and *Rhynchobatus* spp. taken from White et al 2014.

Parameter	<i>Glaucostegus typus</i>	<i>Rhynchobatus</i> spp.
Maximum age	19	12
Age at first pupping	5	6
Fecundity (# female pups)	7	7
Breeding Frequency	annually	annually

Taxonomic and field examination of female reproductive development suggests similar sizes at birth (400 -500 mm L_{ST}) between species within the complex (White and Dharmadi 2007, Compagno and Last 2008). Male *R. australiae* with $L_{ST} > 1300$ mm were found to be maturing and pregnant females were between 2800-3000 mm L_{ST} , with litter sizes ranging from seven to 19 (mean = 14) (White and Dharmadi 2007).

Mortality estimates for *Rhynchobatus* spp. have been limited by poor resolution of fisheries data for non-target species and taxonomic confusion between complex members (White et al. 2013a). Data from semi-quantitative risk assessments and fisheries observer surveys indicate *Rhynchobatus* spp. occur in a range of Australia's northern fisheries (Stobutzki et al. 2002, Salini et al. 2007, Zhou and Griffiths 2008, White et al. 2013a). Observation of the commercial gill-net fishery on the east coast of Queensland revealed *Rhynchobatus* spp. are regularly taken and that individuals of all size classes were encountered in the fishery. Mortality in Queensland waters, however, is limited by management controls, including restrictions regulating the commercial capture and retention of *Rhynchobatus* spp. A commercial fisher is required to have a specific licence that permits retention of a maximum of five individuals per trip and dorsal fins must remain attached to carcasses whilst vessels are at sea.

Glaucostegus typus life history

Biological information for *G. typus* is scarce. Recent examination of *G. typus* from the east coast of Queensland found the species to be relatively long-lived with maximum age estimates of 17 (2450 mm L_{ST}) and 19 years (2500 mm L_{ST}), for males and females, respectively (Table 6.1 White et al. 2014). *Glaucostegus typus* is the largest species of Rhinobatidae, known to reach $L_{ST} > 2700$ mm (Whitely 1939). The largest individual (2840 mm L_{ST}) sampled from the Queensland population was a female estimated at 18 years of age (White et al. 2014). Born at between 380 – 400 mm, *G. typus* reaches sexual maturity, L_{ST} 1500 – 1800 mm (Last and Stevens 2009). Currently no species-specific information is available describing sex ratio of embryos or periodicity of breeding. Litter sizes for smaller Rhinobatidae species (e.g. *Rhinobatos productus*), range from 2 -16 with a sex ratio of 1:1 (Villavicencio 1993 , Márquez-Farías 2007). No mortality estimates are available, but evidence from the assessment of gill-net fisheries in Queensland suggests fishing mortality is generally restricted to juvenile size classes (White et al. 2013a). Previously, larger size classes had been susceptible to trawl gear, however the advent of bycatch reduction devices to all trawl fisheries operating in Australia has likely reduced mortality of larger size classes in this fishery. Currently, licensed commercial fishermen are entitled to take *G. typus* and are not limited by number nor fin attachment laws that apply to *Rhynchobatus* spp.

Sensitivity to uncertainty in life history parameters

To test the sensitivity to uncertainty in life history parameters, scenarios with differing ages of maturity, longevity and natural mortality were constructed for *G. typus* and *Rhynchobatus* spp. (Table 6.2). However, exploration of longevity, natural mortality, fishing mortality and management scenarios were only conducted using life history data for *R. australiae*. Biological information for the complex, although limited, suggests *R. australiae* is the species with the most intermediate biological parameters: larger than *R. palpebratus* and smaller than *R. laevis* (White and Dharmadi 2007, Compagno and Last 2008, White et al. 2014). The species was therefore used as a midpoint for biological parameters within the complex.

Management scenarios

Management scenarios were constructed to examine the effect of size limits on population dynamics under varying levels of fishing mortality ($F = 0.0 - 1.0$). For these scenarios total mortality was the sum of fishing mortality (F) and natural mortality (M). Two maximum size limits were tested, 1500 mm and 2000 mm to account for potential

under-estimates in longevity of *Rhynchobatus* spp. from the life history data used for the models (White et al. 2014). A minimum size limit was also set at 1500 mm, which matched the current Queensland regulation applied to all line caught elasmobranchs. Preliminary biological information suggests *R. palpebratus* are sexually mature and perhaps fully-grown at this length (Compagno and Last 2008), and this reduced the risk of recruitment and/or growth overfishing for this species.

Table 6.2 Scenarios exploring the sensitivity to uncertainty in life history parameters of *Glaucostegus typus* and *Rhynchobatus* spp.

	<i>Glaucostegus typus</i>					<i>Rhynchobatus</i> spp.				
	Age (max)	Age (first birth)	Fecundity	Breeding freq	M	Age (max)	Age (first birth)	Fecundity	Breeding freq	M
Base	17	5	7	Annually	0.24	12	6	7	Annually	0.24
Matures 1 year early	17	4	7	Annually	0.24	12	5	7	Annually	0.24
Matures 2 year early	17	3	7	Annually	0.24	12	4	7	Annually	0.24
Matures 1 year later	17	6	7	Annually	0.24	12	7	7	Annually	0.24
Matures 2 year later	17	7	7	Annually	0.24	12	8	7	Annually	0.24
Longevity *0.8	13.6	5	7	Annually	0.24	9.6	4	7	Annually	0.24
Longevity *1.2	20.4	5	7	Annually	0.24	14.4	4	7	Annually	0.24
Longevity *1.4	23.8	5	7	Annually	0.24	16.8	4	7	Annually	0.24
Longevity *1.6	27.2	5	7	Annually	0.24	19.2	4	7	Annually	0.24
Mortality x 0.5	17	5	7	Annually	0.11	12	4	7	Annually	0.11
Mortality x 0.75	17	5	7	Annually	0.17	12	4	7	Annually	0.17
Mortality x 1.25	17	5	7	Annually	0.29	12	4	7	Annually	0.29
Mortality x 1.5	17	5	7	Annually	0.35	12	4	7	Annually	0.35

Results

Natural mortality

Natural mortality estimates using Jensen's (1996) age independent approach were 0.24 yr⁻¹ for *G. typus* and *Rhynchobatus* spp. Leslie matrix models of unfished *G. typus* and *Rhynchobatus* spp. populations produced generation times of 7.05 and 7.54 years, with net reproductive rates, R_0 of 9.77 and 6.13 female offspring per female newborn, respectively (Table 6.3). Models produced increasing population growth rates, λ , of 1.38 year⁻¹ and 1.27 year⁻¹.

Table 6.3 Effects of variation in maturity, longevity, fishing mortality and exploration of management scenarios for *Glaucostegus typus* and *Rhynchobatus* spp.

	<i>Glaucostegus typus</i>			<i>Rhynchobatus</i> spp.		
	r	R_0	T	r	R_0	T
Base population	0.32	9.77	7.05	0.24	6.13	7.54
Matures 1 year early	0.43	13.28	6	0.18	4.84	8.60
Matures 2 year early	0.60	16.74	4.68	0.24	6.54	7.71
Matures 1 year later	0.25	8.40	8.48	0.08	2.43	10.15
Matures 2 year later	0.12	4.30	11.66	0.04	1.60	10.83
Longevity *0.8	0.32	10.23	7.20	0.14	4.87	10.59
Longevity *1.2	0.32	10.25	7.20	0.14	5.00	10.73
Longevity *1.4	0.32	10.26	7.20	0.15	5.02	10.76
Longevity *1.6	0.32	10.26	7.20	0.15	5.04	10.78
Mortality x 0.5	0.44	27.38	7.51	0.25	10.92	9.53
Mortality x 0.75	0.38	16.05	7.27	0.19	6.16	9.47
Mortality x 1.25	0.26	6.15	6.88	0.07	2.00	9.37
Mortality x 1.5	0.20	3.97	6.72	0.01	1.15	9.31

Sensitivity to uncertainty in life history parameters

Increasing longevity had little effect on rates of population increase for *G. typus*. Variations in natural mortality altered population increase rates with the largest variation evident when mortality rates were reduced: a 50 % reduction in natural mortality

increased population growth rate by 13%. In contrast, a 50 % increase in natural mortality reduced rates of population growth by 17 %. Variations in age at maturity also affected all demographic parameters (Table 6.3). Models in which individuals matured two years later than the base case had the lowest λ of 1.13 year^{-1} and lowest net reproductive rate with newborn females expected to produce 4.30 female offspring during a lifetime. Demographic parameters for *Rhynchobatus* spp. are influenced by increasing and decreasing values for longevity, with 60 % increases in longevity reducing population growth, λ , by 11 %. Increasing natural mortality by 50 % decreased rates of population by 27 % (Table 6.3). Models with earlier ages at maturity had higher rates of population growth $\lambda = 1.27 \text{ yr}^{-1}$ in comparison to models with older ages of maturity $\lambda = 1.04 \text{ yr}^{-1}$ (Table 6.3).

Exploring fishing mortality

Increasing fishing mortality on *G. typus* reduced rates of population increase, with negative population growth occurring at $F = 0.4$ for no size limit and 2000 mm size limit scenarios (Fig 6.1a). Estimates of the generation time (G) also declined under increased fishing mortality with a range of 6.77 to 5.75 years (mean = 6.13 years). Models showed increased fishing mortality of *Rhynchobatus* spp. decreased generation time, 7.42 – 6.74 years (mean = 7.04 years). Increasing fishing mortality reduced intrinsic rates of population increase from $r = .25 \text{ yr}^{-1}$ to 0.01 yr^{-1} and rates of population increase became negative at $F = 0.3$ (Fig 6.1b).

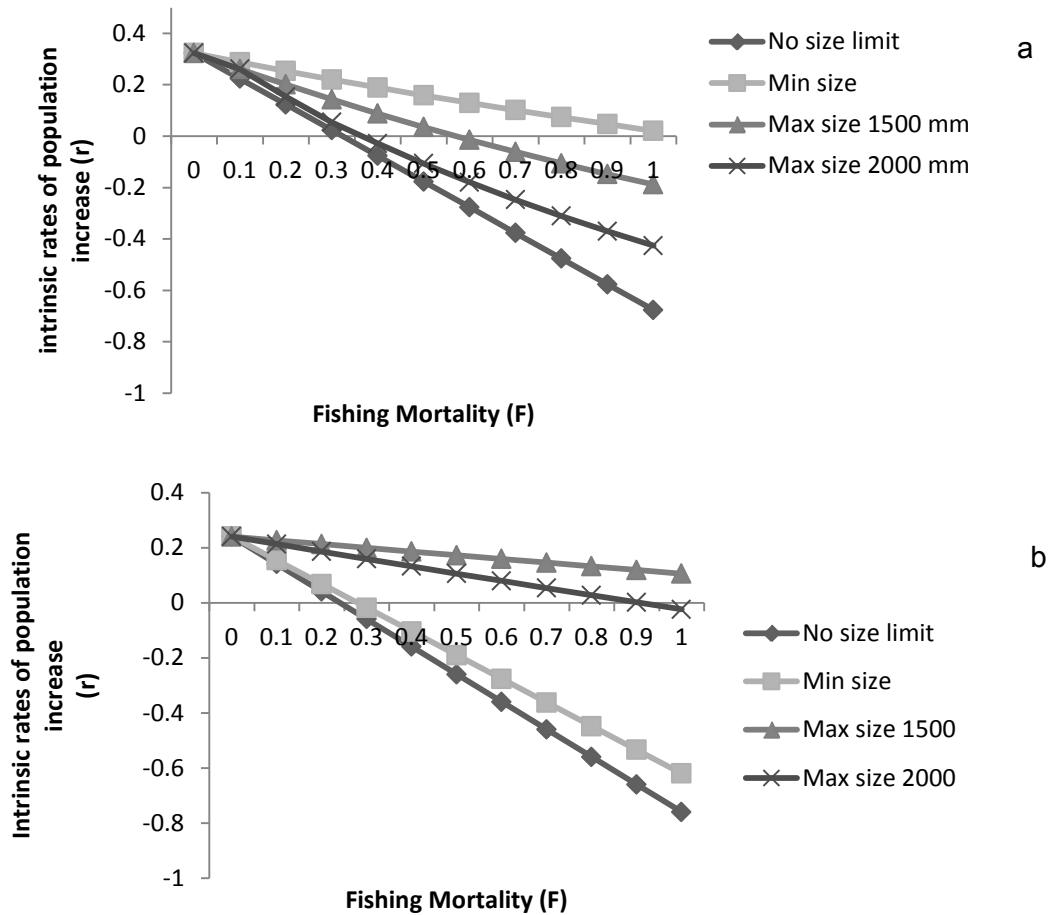


Fig 6.1 Effect of fishing mortality on intrinsic rates of population increase under varying management strategies; a) *Glaucostegus typus* and b) *Rhynchobatus* spp.

Evaluating management options

Exploration of management scenarios found a minimum size limit of 1500 mm had the least effect on the rate of population growth for *G. typus* (Fig 6.1a). Under a minimum size limit regulation, $\lambda > 1 \text{ yr}^{-1}$ for all levels of fishing mortality ($F = 1.0 \text{ yr}^{-1}$ $\lambda = 1.02 \text{ yr}^{-1}$). Rates of population increase declined ($\lambda < 1 \text{ yr}^{-1}$), at both maximum size scenarios (Fig 6.1a). For maximum size limit of 1500 mm $\lambda < 1 \text{ yr}^{-1}$ was reached at $F = 0.6$. At the larger size limit (2000 mm) values declined below those sufficient to sustain population levels at lower rates of fishing mortality ($F = 0.4$). A minimum size limit for *Rhynchobatus* spp. resulted in the lowest rates of population growth with $\lambda < 1 \text{ yr}^{-1}$ occurring at $F = 0.3$. In contrast to *G. typus*, maximum size limits were found to be the optimal management scenario for *Rhynchobatus* spp., providing the closest results to an unfished population

(Fig 6.1b). Both maximum size limits produced $\lambda > 1 \text{ yr}^{-1}$ for all levels of fishing mortality except at $F = 1$ for 2000 mm where $\lambda < 1 \text{ yr}^{-1}$. A maximum size limit of 1500 mm produced the greatest rate of population growth $\lambda < 1.11 \text{ yr}^{-1}$ across all size limits for $F = 1$.

6.3 Discussion

Demographic analysis

Results from this study suggest that unfished populations of *G. typus* and *Rhynchobatus* spp. have rates of population increase sufficient to maintain healthy stocks and generation times that are moderate in comparison to other elasmobranch species. Recent assessments of batoid demography (Simpfendorfer 2000, Dulvy and Reynolds 2002, Frisk et al. 2002), have focused on species thought to be vulnerable to fisheries exploitation (e.g. *Pristis pectinata*, *Dipturus batis* and *Raja clavata*). These studies have revealed that large-bodied, late maturing; long-lived species of batoids, particularly those with temperate distributions are most susceptible to exploitation by fisheries (Dulvy and Reynolds 2002, Frisk et al. 2002). Examination of highly vulnerable species has produced more conservative projections of intrinsic growth and low reproductive potentials with which to compare more resilient species. Models under best-case (e.g. no fishing) conditions for the diamond stingray *Dasyatis dipterura* ($\lambda = 1.06$ Smith et al. 2008), and pelagic stingray *Pteroplatytrygon violacea* ($\lambda = 1.17$ Mollet et al. 2002) produced low levels of population growth in comparison to *G. typus*. Projected population growth rates for *G. typus* were also higher than other species of shark-like batoids such as smalltooth sawfish *Pristis pectinata* ($\lambda=1.14$ Simpfendorfer 2000) and largetooth sawfish *P. perotteti* ($\lambda=1.12$ Simpfendorfer 2000). These findings suggest *G. typus* populations may be less vulnerable to fisheries exploitation than other species of large batoids.

Baseline models for *Rhynchobatus* spp. included all species within the complex, as insufficient data are currently available to segregate fisheries mortality, life history and ultimately population vulnerability. Thus comparisons between projected values using pooled life history data and those for other elasmobranchs would be inappropriate until species-specific information detailing longevity, size at maturity and reproductive productivity can be attained for complex members. Positive rates of population growth reported for the complex may mask species-specific vulnerabilities. Published biological information for the complex suggests large size differences and distinct patterning between species (Whitely 1939, Compagno and Last 1999, 2008, Last and Stevens 2009). However, these have been less obvious during field observations and the complex is managed as a single species in Australian waters. White and others (2014) recently proposed the cautious use of a single growth model for this complex as a guide to

describing life history. Given the relative difficulty of identification and poor resolution of data describing the catch of this species complex by fisheries, continued management at the complex level is most appropriate.

Sensitivity to uncertainty in life history parameters

Model projections revealed age at maturity to be the most influential parameter for both *G. typus* and *Rhynchobatus* spp. populations. There was limited information describing reproductive potential or age/size at maturity for these species and what is available is regionally specific to areas outside Australia (White and Dharmadi 2007). Differences in life history between geographically separated stocks of the same species are commonly reported for elasmobranchs (e.g. Carlson et al. 2003, Driggers et al. 2004, Neer and Thompson 2005, Carlson et al. 2006). Thus, single estimates of life history for species with large geographic distributions may not account for differences between stocks (Carlson et al. 2006). Further work is urgently required to quantify these parameters, both at a species level, but also among regions as even slight variation in maturity and natural mortality rates may have significant implications for the vulnerability of species (Stevens et al. 2000a).

Evaluating management options

Previous qualitative assessments of *G. typus* vulnerability to fisheries have highlighted the distributional overlap of commercial fishing effort with critical habitats of the species (Stobutzki et al. 2002, Zhou and Griffiths 2008). However, recent examination of the commercial gill-net fishery in Queensland, found the size distribution of *G. typus* caught in gill-net gear was skewed toward smaller size classes that represent juvenile and sub-adult individuals (White et al. 2013a). Thus, under current fishing practices in this fishery *G. typus* may exist in a gauntlet fishery (White et al. 2013a), where fishing mortality is restricted to juvenile age classes (Prince 2005). Demographic analysis of *G. typus* suggested a minimum size limit was the optimal management strategy for this species. Minimum size limits are a viable management strategy for avoiding growth overfishing in species with moderate life histories, and have been successfully used in the management of other shark fisheries operating in Australian waters (Walker 1998, Simpfendorfer 1999c, Walker 2004). As a consequence of fishing gear, seasonality of effort (White et al. 2013a) and species spatial ecology (White et al. 2013b), the addition of a minimum size limit would focus mortality to few size classes and likely produce positive outcomes for *G. typus* populations interacting with commercial gill-net fisheries operating under current regimes of gear size and configuration.

Catch data from the Queensland inshore net fishery showed all size classes of *Rhynchobatus* spp. were caught by the fishery (White et al. 2013a). Exploration of demography found that setting maximum size limits for *Rhynchobatus* spp. produced rates of population increase similar to unfished populations. Maximum size limits have been found to be an effective management strategy for some elasmobranchs, particularly those species with high initial growth rates, like *Rhynchobatus* spp. (Walker 2004, White et al. 2014). In Australian waters, legal maximum limits have been applied successfully to shark species in efforts to control mercury concentrations in meat (e.g. *Galeorhinus galeus* Walker 1999) and reduce likelihood of recruitment overfishing (Walker 2004). Careful consideration must be given to the selection of size limits for *Rhynchobatus* spp. Initial taxonomic research suggests *R. palpebratus* is significantly smaller than *R. australiae* and *R. laevis* (Whitely 1939, Compagno and Last 2008, Last and Stevens 2009). With varying maximum sizes, it is likely that size at maturity is also different between species, thus inappropriate size limits may allow for recruitment overfishing, especially of *R. palpebratus*. The setting of a single size limit for a species complex if done with caution is the most appropriate management strategy for regulating catch of *Rhynchobatus* spp. in multi-species fisheries.

Demographic analysis in the present study has revealed that different management strategies were appropriate for *G. typus* and *Rhynchobatus* spp. despite morphological similarities between the two. Variation in life history characteristics and ecology between morphologically similar species is not a new phenomenon among elasmobranchs, and has been particularly evident in investigations of fusiform species (e.g. Carcharhinidae). These findings support the need for species specific data to inform decision making if management strategies are to be successful. Recommendations of minimum size limits are novel for a species with moderate k-select life history strategy. However, current evidence suggests limited fishing mortality of adults in commercial gill-net and trawl fisheries (Stobutzki et al. 2002, White et al. 2013a) and thus these typically vulnerable portions of the population are currently underexploited. Higher growth rates of *Rhynchobatus* spp. in comparison to *G. typus* may be a consequence of both the species complex and limited sample size (White et al. 2014). The application of size limits is further complicated by size variability between species within the *Rhynchobatus* spp. complex. Given higher growth rates and vulnerability to recruitment overfishing setting a maximum size limit was deemed the most appropriate strategy. Differing management strategies between morphologically similar species and within species between size classes/ life history stages have been successfully implemented (Simpfendorfer 1999c, Stevens et al. 2000b, Walker 2004). The success of management strategies will be

dependent upon the continued investigation of species specific biology and ecology so that strategies can be adapted as appropriate.

Sustainability of fisheries

In Australia, targeted management strategies initiated to mitigate anthropogenic impacts on shark-like batoids (fishing activity and habitat degradation) have to a large extent been precautionary and/or indirect. Strategies include precautionary bag limits on *Rhynchobatus* spp., changes to finning regulations for all elasmobranch species and networks of marine protected areas (White et al. 2013a). The use of bag limits for *Rhynchobatus* spp. is of limited value because they offer no protection to recruitment or growth overfishing and may also result in increased unreported take as a consequence of product sorting (Walker 2004). However, implementation of size limits in conjunction with bag limits may reduce fishing mortality, thus reducing the probability that current fishing levels would adversely affect the population.

Unregulated catch of *G. typus* and *Rhynchobatus* spp. throughout their range has resulted in dramatic declines in population size and distribution, particularly in South-East Asia. In these regions, population declines have been driven by the use of gill-net of larger mesh sizes (>20 cm) in comparison to gear used in Australian waters (W. White pers. comm.), and consequently larger size classes (2300–3000 mm) of shark-like batoids are caught (White and Dharmadi 2007). Such population declines demonstrate the vulnerability of these species to fisheries activity, particularly where a combination of demersal set long-line and gill-net gears are used (Bentley 1996b, Chen 1996). The absence of bottom set long-line gear and the inclusion of bycatch reduction devices on trawl gear in Queensland fisheries has likely buffered against population declines associated with fishing pressure, particularly on larger size classes. Size limits are able to be applied to *G. typus* and *Rhynchobatus* spp., as they are able to interact with gill-net gears and survive for long periods, meaning post release survival is likely to be high. Species unable to be retained by fishermen and sold are more likely to be under reported in catch, and are thus harder to manage.

CHAPTER 7

General Discussion



Plate 7. Filming the removal of shark-like batoids from gill-nets. Photo by Jason Stapley (Old Mapoon, 2009).

Significance of this study

There is growing concern for the sustainability of elasmobranchs in fisheries globally (Hoenig and Gruber 1990, Stevens et al. 2000a, Graham et al. 2001, Clarke et al. 2006b, Dulvy et al. 2008). If elasmobranchs are to be effectively managed as target or bycatch species, knowledge of when and where they are available to fisheries is essential to developing sustainable harvest strategies (McAuley et al. 2007a)(Chapter 2). Coastal ecosystems are complex, with dynamic environmental conditions and their close proximity to urban centres means they are often affected by anthropogenic impacts (e.g. fishing, habitat destruction/alteration) (Jackson et al. 2001, Knip et al. 2010, Farrugia et al. 2011). Understanding the spatial ecology of shark-like batoids at both broad (Chapter 3) and fine spatial scales (Chapter 4) is essential to quantifying their vulnerability to fisheries. Additionally, the lack of accurate age and growth information, particularly for non-target elasmobranch species such as shark-like batoids, has been a major stumbling block for fisheries management (Hoff and Musick 1990, Gelsleichter et al. 1998, Ismen et al. 2007). Understanding species biology (Chapter 5) and modelling population dynamics (Chapter 6) provide the final components of assessment necessary for informing management decisions.

Insufficient data regarding the biology of bycatch species and their availability and incidence in fisheries has been a significant hindrance to assessing population viability under fishing regimes, especially for elasmobranchs (Frisk et al. 2001). Amid fears for the long-term stability throughout their range, shark-like batoids have been included in qualitative and semi-quantitative risk assessments of elasmobranch fisheries interactions (Stobutzki et al. 2002, Salini et al. 2007, Zhou and Griffiths 2008). This PhD thesis provides information to help assess shark-like batoid fisheries vulnerability using a quantitative risk assessment. Chapter two provides the first dedicated examination of shark-like batoid interaction with commercial gill-net fisheries in Queensland waters. Results indicated nearshore areas were critical habitat for *G. typus*, supporting a previous semi-quantitative assessment (Salini et al. 2007), that highlighted distributional overlap between shark-like batoids and commercial nearshore gill-net fisheries. However, fisheries independent sampling when combined with fisheries dependent data from observer surveys revealed that large *G. typus* are available to the fishery, but catch was restricted to juvenile and sub adult size classes. Concentrating a fishery on a few juvenile year classes has proven to be a robust management strategy for some elasmobranch fisheries (e.g. gummy shark), particularly for species with low levels of productivity (Simpfendorfer 1999c, Prince 2005, McAuley et al. 2007b, Kinney and Simpfendorfer 2009). The small mesh size of gill-nets currently used in the east coast inshore finfish

fishery (ECIFF) limits the interaction of the ECIFF to smaller *G. typus* and may be a robust example of the usefulness of mesh size restrictions in managing harvest.

The contribution of spatial ecology to management

Reliance on fisheries dependent data for assessing a species' habitat preferences and use is biased by the activity of the fishery. The utilization of multiple survey types in this PhD has overcome issues of fisheries restrictions, spatial regulation and size selectivity of conventional survey gears to provide information on distribution throughout multiple habitat types within the ecosystem.

Baited remote underwater video surveys (BRUVS[®]) proved to be a useful tool in examining the distribution and habitat associations of shark-like batoids, at broad spatial scales. The use of BRUVS[®] provided a non-invasive, non-destructive and minimally disruptive approach that has bridged gaps between historic fisheries records and fisheries independent data. Although morphologically similar, the spatial ecology of *Rhynchobatus* spp. and *G. typus* were different. BRUVS[®] results suggest that marine protected areas (MPAs) may be of limited benefit to mobile habitat generalists like *Rhynchobatus* spp. The success of MPAs has largely been limited to site attached species or those with high site fidelity (Murawski et al. 2000, Galal et al. 2002, Russ et al. 2004). Recent research using BRUVS[®] to quantify reef shark presence in MPAs has revealed benefits of protected zones to several shark species (Bond et al. 2012, Goetze and Fullwood 2013). However, many of the reef shark species examined are known to have high site fidelity and habitat dependence. If *Rhynchobatus* spp. are not resident within discrete areas or habitats then small-scale MPAs may not be beneficial for this group of species.

Current MPAs within the Great Barrier Reef Marine Park (GBRMP) may only afford *Rhynchobatus* spp. limited protection, however, preferences for habitats at depths outside that of current ECIFF operation may isolate portions of the population from current commercial fishing activity. Given the current conservation concern for *Rhynchobatus* spp. and greater incidence within the ECIFF in comparison to *G. typus*, identification of habitat preference outside of current commercial fishing effort is an important finding of this dissertation, and potentially a key component in the resilience of these populations to fishing exploitation in Queensland waters. Acoustic monitoring data supported BRUVS[®] findings that *Rhynchobatus* spp. are habitat generalists, however it is unclear whether there are species-specific preferences of habitat use within the complex.

The utility of BRUVS[®] was severely limited by high turbidity and considered unsuitable for quantifying spatial ecology of *G. typus* that prefers turbid habitats. Acoustic telemetry was

utilized in these habitats to examine use of space and residence in nearshore areas. Similar to BRUVS[®] data, long-term acoustic monitoring revealed that despite being morphologically similar *G. typus* and *Rhynchobatus* spp. exhibited different spatial ecology. *Rhynchobatus* spp. tended to be present for longer continuous periods in Cleveland Bay, while *G. typus* were present for shorter, predictable periods (White *et al* 2014). Inter-annual consistency in activity space size and location within nearshore areas, coupled with the reproductive stage of individuals suggest Cleveland Bay provides critical habitat for *G. typus*. Philopatry and site attachment of adult *G. typus* suggests marine protected areas within nearshore regions could offer benefit to these species during periods of coastal utilisation. Additionally, existing temporal closures regulating ECIFF activity (e.g. *Lates calcarifer* November – February) and low incidence in the ECIFF despite availability due to fishing practices, may already be providing significant protection to *G. typus* populations in Queensland waters.

Role of life history information

Data deficiency, especially in terms of life history, has been a major impediment to the management of many elasmobranch species (Simpfendorfer *et al.* 2011a). Despite growing exploitation of shark-like batoids, published information describing their basic biology is scarce. This dissertation provided estimates of age and growth for *G. typus* and *Rhynchobatus* spp. von Bertalanffy growth models have traditionally been used in the estimation of elasmobranch growth curves. However, these models do not account for changes in growth rate associated with changing life history stages (e.g. energy allocation to reproductive development or gestation), or environmental conditions (changes in temperature, and prey availability) (Araya and Cubillos 2006, Braccini *et al.* 2007, Dale and Holland 2012). There is increasing support for the use of sigmoid growth functions in elasmobranchs (Dale and Holland 2012), particularly for species where growth differs between distinct stages of life history (Carlson and Baremore 2005, Braccini *et al.* 2007). Sigmoid growth functions (Gompertz and logistic) best described the growth of *Rhynchobatus* spp. and *G. typus*, providing both the most biologically palusible growth model and the best statistical fit. Given that sigmoid growth functions were preferable in this instance, the extent of comparison of life history derived for other elasmobranchs using von Bertalanffy growth functions is limited. Further work is required to resolve the life history of other batoid species to assess species-specific vulnerability to exploitation. Such work provides valuable data for informing decision management tools and identifying species of priority for conservation initiatives.

Until species-specific identification is possible defining the life history of *Rhynchobatus* spp. will remain complicated. Variability of length with increasing age observed here may reflect variability in growth between the smallest species (*R. palpebratus*) and the larger bodied *R. australiae* and *R. laevis*. Further work is required to resolve taxonomy of the group and define species-specific life history parameters. Taxonomic confusion of elasmobranchs is not uncommon with many sharks, particularly those from the carcharhinid family being difficult to identify to species level using only morphological features (Chan et al. 2003). Recent work by Harry *et al.* (2012) and Morgan *et al.* (2012) has highlighted the importance of using multiple lines of evidence (ecological, morphological and molecular) to distinguish between groups of closely related and cryptic species rather than reliance on a single method (e.g. external morphology, genetics or vertebral counts). Validation of current species classification and validation of species-specific life history information for the *Rhynchobatus* spp. complex will require multiple lines of enquiry.

Implications for fisheries management

Current management strategies in this data deficient environment are either precautionary (e.g. bag limits for *Rhynchobatus* spp.) or indirect (e.g. use of bycatch reduction devices (BRD) by trawl fisheries; implementation of marine protected areas). Historically shark-like batoids have been of low economic value (Chen 1996) and therefore, there is little baseline data on landings, fishing related mortality and un-fished biomass in Australian waters (Stobutzki et al. 2002, Salini et al. 2007). Issues with species identification for the *Rhynchobatus* spp. persist and under reporting of landings for all species continue to inhibit efforts for determining population status. There are currently no restrictions on the number or size of *G. typus* that can be taken by commercial fishermen operating in the ECIFF with an endorsement for retaining elasmobranchs. Bag limits currently apply in Queensland waters for *Rhynchobatus* spp. to any commercial fishermen with an endorsement for elasmobranchs. However, this output control may result in product sorting whereby smaller individuals retained by the fisher are disposed of so that larger, more valuable individuals can be marketed. Under these circumstances, cryptic mortality may rise, reducing the ability to accurately assess impacts of fishing activity. Population sizes of the individual species may have been overestimated because *Rhynchobatus* make up a complex and not a single species. It follows, therefore, that the effect of fishing on individual species within the complex may have been underestimated. The implementation of size regulations in conjunction with the existing bag limits may help to mitigate recruitment overfishing. Resolution of species-specific life history data may

facilitate the introduction of effective management strategies that would protect the complex based on the species with the most vulnerable life history.

The combination of fisheries interaction, demographic modelling and spatial ecology produced a comprehensive overview of *G. typus* and *Rhynchobatus* spp. biology and ecology. Despite being morphologically similar *G. typus* and *Rhynchobatus* spp. exhibited differences in both biology and spatial ecology. Previously assessed as high risk due to distributional overlap with inshore gill-net fisheries, this PhD found *G. typus* has limited interaction with the ECIFF, and exhibits strong site fidelity and philopatry to nearshore areas. Well placed marine park zones within nearshore areas would provide significant protection to aggregations of sexually mature individuals. Clear definition of biology and spatial ecology of *Rhynchobatus* spp. is complicated by the species complex. However, preferences for deeper water habitats outside the current commercial gill-net fishery may afford this complex considerable protection from exploitation. Both long-term monitoring acoustic data and BRUVS[®] found *Rhynchobatus* spp. to be mobile habitat generalists, raising questions about the utility of MPAs for conserving these species. Species-specific data describing both biology and spatial ecology of *Rhynchobatus* spp. is required as previous assessments were derived based on the assumption of a single species in Queensland waters, rather than the three species complex. High conservation concern for shark-like batoids has been driven by depletions in South-233 East Asia where large mesh sized gill-nets and bottom set long-lines are used to target these species. Currently there are no target fisheries for shark-like batoids in Australian waters, however given the high value of their fins they will remain valuable byproduct species. The occurrence of shark-like batoids in mixed species fisheries complicates the task of management, however the evidence provided here suggests that under current fishing regimes and management strategies the populations may be sustainable.

Future Directions

This dissertation is the most comprehensive assessment of shark-like batoid biology and spatial ecology to be conducted in Australian waters. Throughout the dissertation *R. australiae*, *R. laevis* and *R. palpebratus* have been treated as a species complex and this is how they are currently managed in Australian fisheries. However, previous assessments of other elasmobranchs have found significant variance in life history between species within families (e.g. Carcharhinidae, Sphyrnidae). Further work is therefore required to resolve species specific spatial ecology and life history for members of the *Rhynchobatus* spp. complex. For practicality, these species may continue to be

managed as a complex, but strategies should be developed to protect the most vulnerability species.

Nearshore area were identified as being critical habitat for *G. typus*, however where they go when they left the monitored area remains unclear. Evidence from BRUVS[®] survey suggests *G. typus* prefer nearshore habitats and do not commonly use other habitats. These habitats are amongst the most impacted marine habitats in Australian waters. Further work is required to understand how *G. typus* use these areas across broad spatial scales so interaction with anthropogenic impacts can be assessed and mitigated if necessary.

Rhynchobatus spp. was found to be a habitat generalist, broadly distributed throughout a range of habitats within the ecosystem. Further research is needed to quantify species-specific spatial ecology. Rather than a habitat generalist, there may be distinct habitat preferences and seasonal movement that are masked by the presence of three such morphologically similar species. Defining depth preferences of *Rhynchobatus* spp. below those at which the ECIFF currently operates was one of the most significant findings of the dissertation. As a complex it suggests there is a common preference for deeper habitats among species. Whilst the present study improves our understanding of *Rhynchobatus* spp. spatial ecology, particularly in nearshore waters further work is required to assess whether distinct habitat preferences exist between species, and whether as is the case with *G. typus* there are critical habitats and seasons when individuals aggregate to reproduce (e.g. mating and/or pupping). Further definition of spatial ecology for all complex members will allow targeted, species specific management initiatives to be developed to manage the interaction of *Rhynchobatus* spp. with commercial fishing activity within the GBRMP.

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