21st Century Small-scale Fisheries of Belize: A Legacy of Eroding Size-Spectra, Trophic Shifts and Underinvestment in Fishery Management

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Abstract

After 100 years of commercial fishing, the coastal waters of the globally significant Belize barrier reef complex are showing signs of overfishing, food web compromise and threats to biodiversity. These challenges are exacerbated by multiple ecosystem stressors including climate change and nutrient enrichment. The Government of Belize has had some success in facilitating ongoing fisheries production and increasing exports through large and long-term investments in an internationally recognized marine protected areas (MPAs) network, including a UNESCO World Heritage site, and banning of several gear types (trawlers, gillnets, air-supply systems). However, increasing fishing effort has resulted in significant removals of juveniles across a broad swathe of species. While the scale and complexity of the co-managed MPA system is impressive, the sheer scale of fishing impacts may render MPAs insufficient as the primary tool for their mitigation. Given finance limitations, blue bonds, via private sector investment, are being used to promote ocean and coastal area conservation, most notably through large expansions of no-take and multiple use MPA areas. The success of this approach will be measured using common, but overly simplified evaluation metrics (e.g., area conserved) despite limited local or international evidence that these metrics equate to conservation impact. In addition, the use of more traditional, species-specific fisheries sustainability policies, including minimum sizes and fishing bans for a number of large, reef-obligate groupers, snappers and sharks, have largely not been implemented nor specifically proposed even as temporary measures to allow stocks to rebuild. Such approaches should take place under well considered species management plans as mandated in the most recent Belize Fisheries Act. The lack of such actions over decades has contributed to a steady erosion of marine populations with increasing negative consequences for fisher livelihoods (3000+), associated processing jobs (15,000+), food security for a population of 400,000 and benefits from over half a million visitors annually.

Keywords: small-scale fisheries; coral reef; marine protected areas; management; biodiversity; invertebrate; bony fish; elasmobranchs; livelihoods; food security

1. Introduction

Small-scale fisheries (SSF) engage approximately 97% of the world's fishers, and are responsible for half of the world's fish production (Andrew et al. 2007, Kelleher et al. 2012, Vianna et al. 2020). The vast majority of SSF operate from shore or from small boats (<20 m in length), targeting invertebrates, bony fish, and elasmobranchs across a complex of shallow, inter-connected tropical marine habitats dominated by mangrove, seagrass, sand flats and coral reefs. Globally, these SSF are critical in facilitating livelihoods and food security as well as providing important exports and foreign currency revenue to millions of people in many developing nations (Andrew et al. 2007, Béné et al. 2010, Anderson et al. 2011). Despite their importance, they are not as well studied nor understood as industrial fisheries, and operate very differently. They are typically characterised by multi-species and multi-gear extraction dynamics, numerous small and geographically dispersed landing sites, limited processing facilities, and fragile transport chains with often inadequate data available for use in conventional stock assessments based on population dynamics models (Costello et al. 2012, Babcock et al. 2018, Tewfik et al. 2022) – all of which makes their management particularly challenging.

The impacts of human population growth on earth's resources have increased the complexities of SSF management. Increasing catches, often illegal, unregulated, or unreported, have resulted in significant declines in mature biomass for many species across habitats along with a loss of biodiversity and ecological function (Pauly et al. 1998a, Pauly and Zeller 2016, Cinner et al. 2020). This has been exacerbated by nutrient enrichment from agricultural and urban development and global climate change impacts on temperature, calcification, and primary productivity (Steffen et al. 2015, Malone and Newton 2020, Davis et al. 2021, Bieg et al. 2024). As the fraction of landed individuals that are mature declines (earliest and most acutely for larger species), future recruitment is threatened and target size-spectra (the relationship between organism size and abundance) shifts to smaller, lower trophic level species with lower market value, and may result in ecosystem wide trophic cascades (Goni 1998, Pinnegar et al. 2000). This has ecological and socio-economic implications, resulting in ongoing declines in fishable biomass, biodiversity and trophic stability as well as losses in labour opportunities, cash income and food security. These impacts are particularly acute for resource-poor households and communities, which tend to be directly reliant on ecosystem services for livelihood and well-being, and limited in their ability to adapt when these services are lost (Pauly et al. 1998b, Jackson et al. 2001, Valentine and Heck 2005, Rooney et al. 2006, Béné et al. 2010).

The dynamics of overfishing are evident in Belize, which is situated at the heart of the globally significant Meso-American Reef complex. While Belize has invested heavily into the designation and management of marine protected areas over the last 40 years in support of fisheries and conservation (Gibson et al. 2004, McField et al. 2024a), the lack of appropriate investments into other traditional species-specific fisheries sustainability policies for most species (e.g., species size regulations) has led to the general erosion of exploited populations (Sala et al. 2001, Acosta 2006, Graham et al. 2008, Babcock et al. 2018, Tewfik et al. 2020, Tewfik et al. 2022). This review examines the patterns of change observed in critical SSF in Belize against the backdrop of historical extractions, existing management practices and ongoing reef complex habitat degradation.

2. Background

2.1 A Brief History of Fishing in Belize

There is a long history of marine animal harvest throughout the Caribbean, including in the area of Belize, illustrated by archeological materials and accounts from colonial records spanning at least 1,500 years. This is not surprising given the biodiverse and productive nature of the shallow mangrove-seagrass-coral reef complex at the heart of the Meso-American reef (Heyman and Kjerfve 2001). Evidence from middens indicates that Mayan coastal fishing outposts consumed queen conch (*Aliger gigas*), Caribbean spiny lobster (*Panulirus argus*) and many large reef fish (Lange 1971, McClenachan et al. 2024) and were dependent on seafood due to the unsuitability of coastal lands for agriculture (Craig 1966, McKillop 1984, Cunningham-Smith 2011). These marine resources were also supplied to large inland Mayan cities including Caracol and Lamanai, both in present day Belize, as well as Tikal in present day Guatemala (Cunningham-Smith 2011). During much of the pre-colonial period, extraction levels were likely lower than today but still significant enough to begin modifying animal populations and ecosystem function (Jackson 1997, McClenachan et al. 2010). Indigenous people continued to make use of abundant marine resources well into the colonial period (post-1600) and long after many other Mayan urban centers were abandoned. During the colonial era, Europeans and enslaved Indigenous and African people also used a variety of marine resources for subsistence for centuries due to the ease of extraction in nearby shallow habitats. The significance of subsistence use of marine resources in Belize persists in the modern age (Zeller et al. 2011, Palomares et al. 2023) and was perhaps best exemplified during the global COVID-19 pandemic when foreign tourism revenues and associated livelihoods dropped significantly (Bennett et al. 2020, Higgs 2021). English colonists in the 17th to 19th-centuries focused on the commercially profitable netting of marine turtles and harpooning of manatees and the Caribbean monk seal (*Neomonachus tropicalis*) to supply large sailing crews that traded between Europe and the Americas (Craig 1966, Jackson 1997). The intense use and trade in marine turtles, including eggs, and manatees, which persisted into the mid-20th century, contributed to depletion of these large vertebrate species to a small fraction of their historic numbers and the extinction of the monk seal (Jackson 1997, McCauley et al. 2015).

The modern commercial exploitation of marine resources in Belize may be traced to the early 1900s with the development of a sturdy sailing craft design (“smack” boats), originating in Cuba and still used today, and focused on highly profitable and abundant species including spiny lobster, queen conch, groupers and snappers (Craig 1966). The modern commercial lobster fishery emerged in the northern cayes of Belize in the 1920s with the introduction of a lobster trap from Canada (Craig 1966). While initially limited to local consumption, commercial fisheries for both spiny lobster (*P. argus*) and queen conch (*A. gigas*) fisheries grew regionally in the 1930s and increased further following World War II (Bene and Tewfik 2001). This growth was spurred by the continuing development of traps for lobster as well as the introduction of outboard engines, basic processing facilities, refrigeration technology and increasing export opportunities (Price 1987, Huitric 2005). By the 1960s, the two principal invertebrate fisheries integrated hook-sticks and free-diving techniques as they expanded to more distant grounds across much of the "shallow" waters (<30 m) of the Belize barrier reef complex (Huitric 2005, Tewfik et al. 2020). Queen conch and spiny lobster remained the main exported commodities since the start of data collections by Food and Agriculture Organization of the United Nations (FAO) in 1950 with conch and lobster expanding significantly in the late 1970s and into the beginning of the 21st century (Fig. 1).

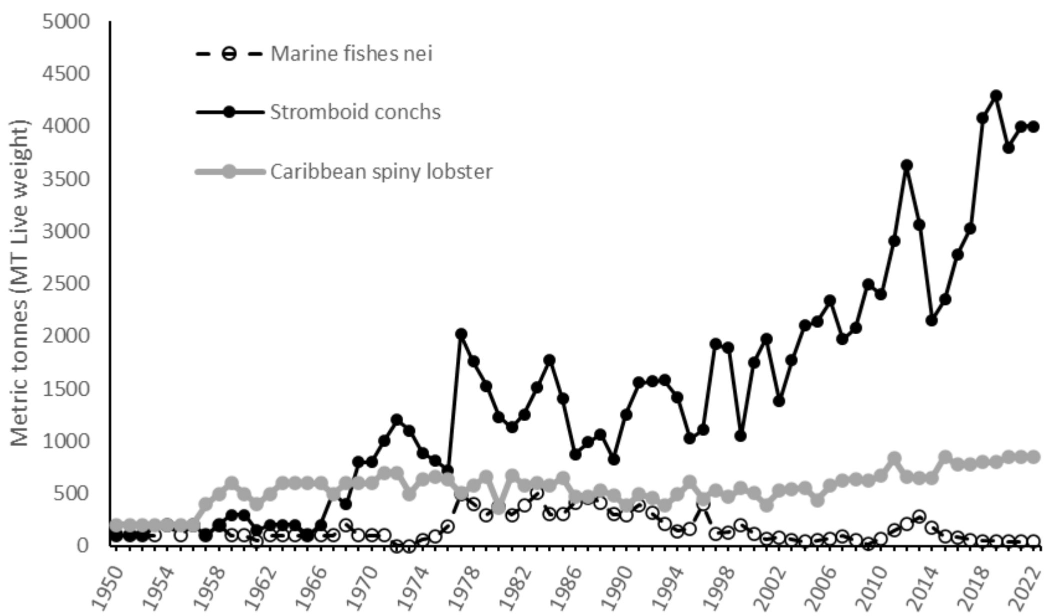


Figure 1: Marine fish (nei – not elsewhere indicated), queen conch (reported as *Strombus/Lobatus gigas*) and Caribbean spiny lobster (*Panulirus argus*) catches from Belize from FAO Area 31 (Western Central Atlantic) between 1950 and 2022 from the global capture production database (FAO 2023). Note that all export values are estimated by FAO for the period of 2017 – 2022 due to lack of reporting by the Government of Belize. Reef fish taxa (e.g., groupers, snappers, grunts) are not tracked separately and combined as marine fish nei. Queen conch biomass in FAO data includes the shell (approx. 90% of total mass). Conch meat is removed from the shell on the fishing grounds and rarely landed. Conch meat exports should be interpreted as approximately a tenth of the reported live weight values in the capture production data (which include the shell) making lobster the largest exported seafood commodity in Belize. A small amount of conch shell is exported for the curio trade and tracked, as with all conch products (e.g. opercula, added value products) in CITES export permits and available in the CITES database (https://trade.cites.org/).

The commercial extractions of groupers, most notably Nassau grouper (*Epinephelus. striatus*), formed the bulk of documented reef fish extractions in Belize by the middle of the 20th century (Craig 1966, 1969, Phillips et al. 2025). This was dominated by regular and large-scale activities at several multi-species fish spawning aggregation sites (FSAs) including Caye Glory (a.k.a. Emily) beginning in the mid-1920s where the smack boats allowed small crews to access remote sites for extended fishing trips around Nassau grouper spawning aggregations on the full moon in December and January each year. Catches of Nassau grouper, the most caught species in Belize, continued to increase over the next three decades with more than 300 boats converging on Caye Glory targeting an aggregation estimated at 100,000 fish during the 1964 - 1965 season (Fig. 2) (Craig 1969, Carter et al. 1994, Heyman and Wade 2007).

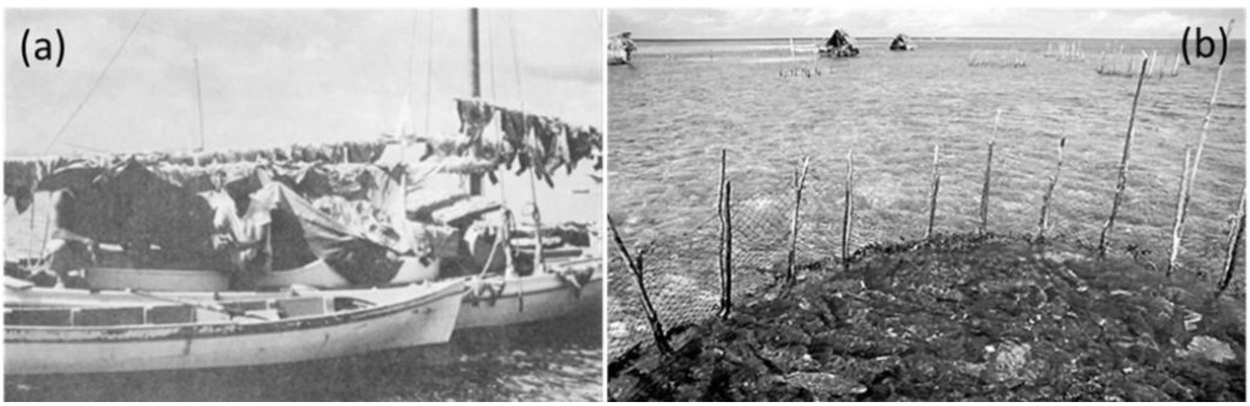


Figure 2: (a) Belizean "Smack" boat with drying Nassau grouper used to transport supplies and conduct fishing at the Caye Glory spawning aggregation (Craig, 1969), (b) Live Nassau grouper holding pens and fisher shelters (background) at Caye Glory (Paz & Truly 2007).

The introduction of the spear gun in the late 1960s likely increased fishing on Nassau grouper during non-spawning periods as with many other large reef fish populations around the world (Sluka and Sullivan 1998). By the time of Belize's independence in 1981, decades of unmanaged extractions had caused acute declines in Nassau grouper landings (Carter et al. 1994, Sala et al. 2001) and fishers had transitioned to other large, aggregating groupers and snappers, simply recorded as "mixed groupers" or "scalefish" in national records (Paz and Sedberry 2008). Marine fish exports in FAO statistics do not differentiate between common reef complex taxa such as groupers, snappers, jacks, and grunts but mixed reef fish landings increased significantly overall by the mid-1970s (Fig. 1) (FAO 2023). Lacking national statistics, species specific trends of finfish can only be derived from local observations. Historically fishers also targeted the smaller aggregations of black grouper (*Mycteroperca bonaci*). These aggregations occurred at the same sites, but after the peak spawning season for Nassau grouper (Dec – Jan) in Belize, appearing in moderate numbers in February and March (Paz and Sedberry 2008). The goliath grouper (*Epinephelus itajara*) and cubera snapper (*Lutjanus cyanopterus*), two of the largest reef fish in the central Western Atlantic region (FAO Area 31), had also been traditionally harvested by coastal fishers in Belize for decades, including at FSAs, using a variety of gear (Heyman and Graham 2000). By 1986 estimates of Nassau grouper at the Caye Glory FSA had declined to less than 20,000, a fifth of the numbers recorded two decades earlier (Carter et al. 1994, Heyman and Wade 2007). An assessment in the early 1990s indicated that Belizean finfish stocks overall were moderately exploited when compared to the heavily exploited waters of Jamaica (Hardt 2009). This was based upon the availability of "prime commercial species" (snappers, and groupers) that were the main source of an export-oriented fishery (Koslow et al. 1994). At the time of inscription of the "Belize Barrier Reef Reserve System" UNESCO World Heritage site in 1996, the IUCN evaluation reported that, "Commercial fish stocks are declining as stocks of many species have been over-exploited" and that "Catches of conch and lobster have significantly dropped over the past decade" (Byron and Osipova 2013).

Belizean fishers also engaged in regular extraction of sharks for decades, with local fishing cooperatives purchasing these products for export from the 1960s to the 1990s (Graham 2007). Up to 3.5 metric tons were exported annually as dry salted fillet to neighboring countries, registered for administrative purposes by the Department of Fisheries, between 1993 and 1995 (Gibson et al. 2004). These fishing cooperative purchases ceased in the late 1990s following a dramatic decline in catch rates in the late 1980s and early 1990s. Investigations at ten coastal communities in Belize revealed that catches of reef-associated elasmobranchs that were once common had declined dramatically over the 20 years following the introduction of gill nets in the 1970s. This decline matched export data from cooperatives (Graham 2007) but is not well reflected in FAO data, with only small contributions noted (FAO 2023).

2.2 Fishing Regulations and Marine Protected Areas

Prior to the 1950s, marine fisheries were primarily for subsistence and local commerce and were not officially regulated, with national exports accounting for less than 1% of production (Thompson 1944, Karlsson and Brycesson 2014). In 1948, the Fisheries Ordinance set the framework for future fisheries regulations, and a Fisheries Officer was appointed to collect data and monitor stocks. A Fisheries Unit (under colonial rule) was established a year later (Huitric 2005). By the late 1950s, the lobster industry was considered established by the colonial government and export duties were applied (Bradley 1956). A number of other regulations pertaining to the local purchase and export of conch and lobster products were established with the increasing influence of fishing cooperatives (e.g., purchase from members only) over foreign buyers and exporters.

The first species-specific extraction regulations in Belize were enacted by the Fisheries Unit in 1977, shortly before independence. These regulations included queen conch minimum legal sizes for shell length and "market clean" (a processed semi-fillet) meat weight, as well as a closed season and were in direct response to signs of rapidly declining catch (Gibson et al. 1983). At the time of enactment, it was believed that these regulations would be refined after a sufficient period of research on the species and monitoring of the effect on the industry (Gibson et al. 1983, Strasdine 1988). However, these regulations have remained unchanged as a minimum shell length of 178 mm and a “market clean” meat mass of 85 g as well as a seasonal closure from July through September (Singh-Renton et al. 2006, Anonymous 2013, Tewfik et al. 2019).

A number of other fisheries regulations have been put in place by the government of Belize in the intervening years. These include: a ban on the use of SCUBA and surface supplied air systems to collect any seafood; closed seasons for spiny lobster (recently amended, March 1st to June 30th) (Anonymous 2021) and Nassau grouper (1 December − 31 March); and size limits for spiny lobster (*P. argus*) (recently amended to >83 mm carapace length (CL) without a correlated change to tail mass but with the latter increased to >128 g) (see section 4.2) (Wade et al. 1999, Anonymous 2021). Spiny lobster traps require escape gaps of 54 mm (2 and 1/8 inches) to limit undersized capture and biodegradable panels to avoid ghost fishing. In addition, regulations pertaining to Nassau grouper (*E. striatus*) include a minimum and maximum total length (50−76 cm) and the provision to be landed whole. All other fish species landed as fillet must include a skin patch (5 × 2.5 cm) suitable for species identification in order to prevent circumvention of the whole Nassau grouper landing rule (Phillips et al. 2025). However, the genetic examination of individual fillets in local markets (2009 – 2011) revealed that 32 - 51% were mislabeled species (Cox et al. 2013). New regulations in 2021 mandate that: (1) shark fishing can only be done by those with a special license; (2) sharks may not be landed on deck within a 3.2 km (2 mile) radius of any of the three atoll marine reserves (Lighthouse, Turneffe and Glover’s); (3) sharks must be landed whole to prevent finning; and (4) a closed season for shark fishing is imposed from May to October (Anonymous 2021).

Trawlers (2011) and gillnets (2020) have also been banned (UNCTAD 2022). The most recent Fisheries Act (Anonymous 2020) also gives a list of fish species prohibited from capture at anytime (Section 88, pg 150): whale shark (*Rhincodon typus*); nurse shark (*Ginglymostoma cirratum*), sawfish (*Pristis perotteti*, P*. pectinate*); all species of rays (Batoidea); all parrotfish (Scaridae); all surgeonfish (Acanthuridae); all angelfish (Pomacanthidae); and all triggerfish (Balistiade). Most of these species are not traditionally caught or have a low rate of extraction reported in landings records (Heyman and Graham 2000, Tewfik et al. 2022). Possible exceptions include nurse sharks before their protection in 2020 (UNCTAD 2022) and parrotfish, which increased in catch frequency from 6% in 2004 to ~20% by 2008 at Glover’s Reef before their ban in 2009. Stoplight parrotfish (*Sparisoma viride*) was the second most caught species between 2004 and 2011 with their increased availability linked to a broader trophic cascade (Anonymous 2010, Mumby et al. 2012, Babcock et al. 2013). Despite the ban, parrotfish abundance data collected over two decades show declines in both fished and protected areas of Glover’s Reef (McClanahan and Muthiga 2020). A number of species important to recreational catch-and-release fishing associated livelihoods (i.e., tourism) - permit (*Trachinotus falcatus*), tarpon (*Megalops atlanticus*), and bonefish (*Albula vulpes*) - may not be landed, with licencing for these activities being separately managed by the Belize Coastal Zone Management Authority and Institute (UNCTAD 2022).

Finally, all fisheries in Belize are managed holistically, in principle, through a large investment in an increasing network of marine protected areas, which include marine reserves (e.g., Glover's atoll) and embedded replenishment zones (i.e., no-take areas), wildlife sanctuaries (e.g., Swallow Caye), national monuments (e.g., Great Blue Hole) and fish spawning aggregation sites (FSAs), with the first MPA established at Hol Chan in 1987 (Gibson et al. 2004, Tewfik et al. 2017, UNCTAD 2022) (Fig. 3). FSAs designed primarily for the protection of Nassau grouper include 11 fully protected sites (2003 Statutory Instrument 161) and two additional sites where a special licence may be granted for "traditional" extraction of Nassau grouper (2003 Statutory Instrument 162) (Fig. 3). In 2016, the nationwide network of Territorial Use Rights for Fishing areas (TURFs, 8 licence specific areas and a 9th common deep-water area) was established and superimposed onto the existing marine protected area system (Fig. 3). All of these spatial management regimes and associated TURF licences for approximately 3000 fishers are overseen by the Belize Fisheries Department under the newly formed Ministry of Blue Economy and Civil Aviation (Karr et al. 2017, Tewfik et al. 2020). Finally, under the latest Blue Bond debt restructuring initiative (Belize Sustainable Oceans Plan; BSOP) the Government of Belize is committed to designating additional areas for conservation (Chakalall and Tsuneki 2021) with an expansion of "Biodiversity Protection Zones" (Fig. 3) to 30% by November 2026 (Anonymous 2024). This will include 15% high protection zones that are not suitable for extraction or sea-bed alteration and considered as no-take but where scientific research and sport-fishing may occur. An additional 15% will be classified as medium protection zones where habitats and species are deemed to be tolerant of some disturbance and sustainable human activities. The latest designation of new "Biodiversity Protection Zones" include significant areas of deep water, which do not protect the most productive reef complex habitats and associated traditional shallow-water (< 30 m) fisheries for queen conch, spiny lobster, groupers, snappers and other reef associated species.

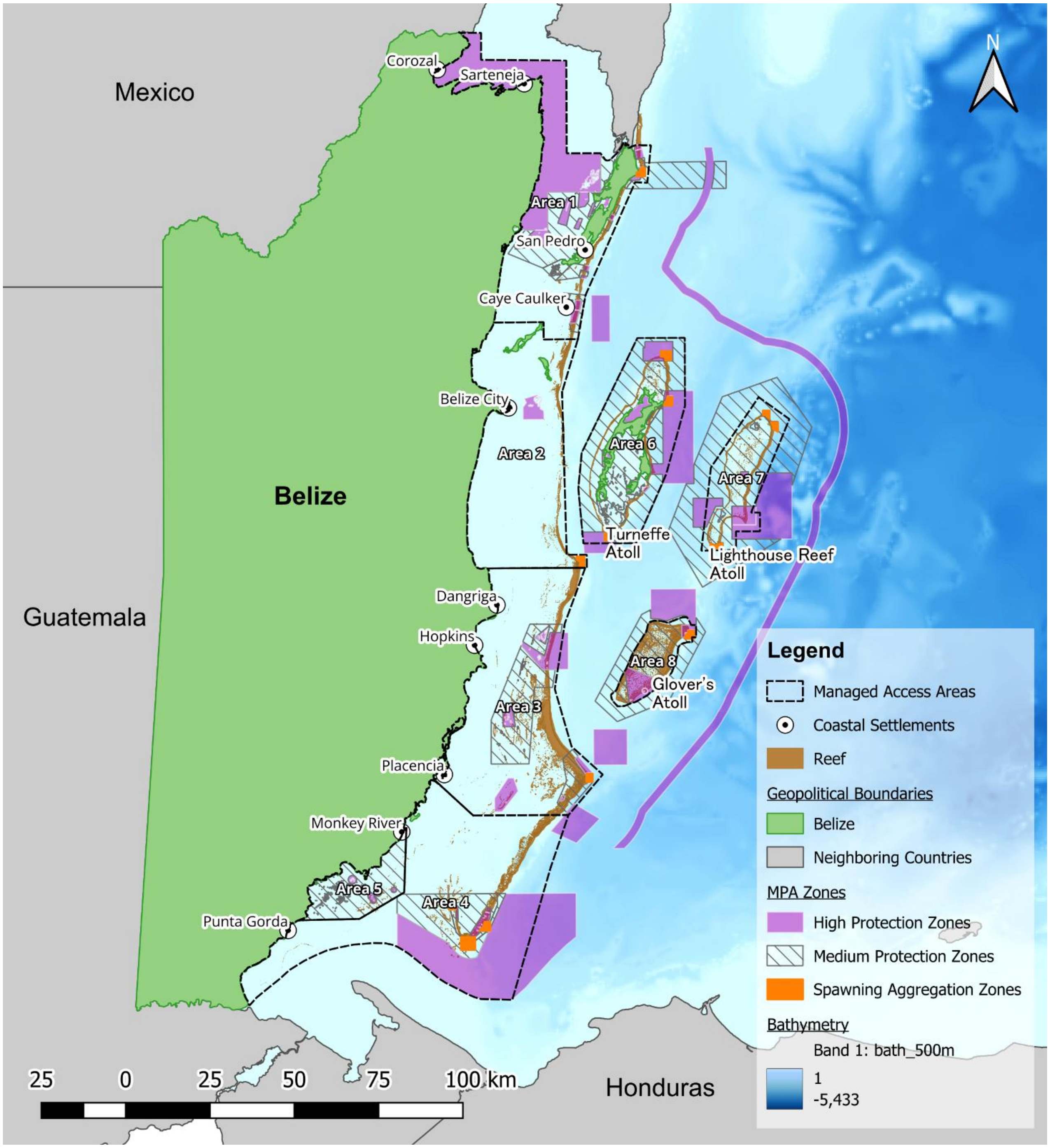


Figure 3: Distribution of Marine Protected Areas (MPA), Fish Spawning Aggregation (FSA) zones & Territorial User Rights for Fishing (TURF) areas in Belize. Recent and proposed expansions of marine reserves to high and medium protection for biodiversity zones are included and detailed in Anonymous (2024).

2.3 Declining Health of Reef Complex Habitats

The warm-water reef complex comprises interconnected mangrove, seagrass, sand flat and coral habitats. It supports much of the world's marine biodiversity, provides significant biomass production for fisheries-based livelihoods and protects coasts from cyclonic storms, which are increasing in frequency and severity due to climate change (Gardner et al. 2003, Hughes et al. 2003, Bellwood et al. 2004). However, these ecosystems, once defined by highly rugose living coral reefs, have been in significant decline since the 1970s due to a growing number of recognized stressors including overfishing, nutrient enrichment, increases in various algae (i.e. "coral-algae phase shift"), coral diseases, and climate change (Arias-González et al. 2017, Lapointe et al. 2021, Bieg et al. 2024), and are now considered one of the most threatened ecosystems on our planet (Birkeland 2004, Hughes et al. 2017). Branching (e.g., *Acropora* spp.) and boulder (e.g., *Orbicella* spp, *Montastrea* spp.) corals (Fig. 4) accounted for nearly 90% of the reef framework over thousands of years but have recently under gone significant losses (Toth et al. 2019, Alvarez-Filip et al. 2022). Overall hard coral cover declined from ~50% to ~10% over the last 50 years in the Caribbean basin (Gardner et al. 2003, Jackson et al. 2014, Lapointe et al. 2019), and "weedy" (stress-tolerant, relatively rapid growing, early-successional) coral species (e.g., *Porites* spp., *Undaria* spp.), lacking structural bulk and complexity, have now become the main components of diminishing hard coral cover, including in Belize (McClanahan and Muthiga 1998, Jackson et al. 2014, McClanahan and Muthiga 2020). The overall changes in reef structure have also resulted in a significant loss in reef complexity and associated ecological function (Alvarez-Filip et al. 2009, Pittman et al. 2011). In the wake of these losses, competition to hard corals and "soft" structure by sponges, including the giant barrel sponge (*Xestospongia muta*), has been increasing (Loh et al. 2015, McMurray et al. 2015, Tewfik and Loh 2017). A recent review indicates that warm-water coral reefs are virtually certain (>99% probability) to reach a thermal tipping point within the next ten years where changes become self-perpetuating and difficult to reverse (Lenton et al. 2025).

Figure 4: Typical reef complex habitats in Belize featuring (clockwise from top-left) nurse shark (*Ginglymostoma cirratum*) on patch reef, tiger groupers (*Mycteroperca tigris*) at spawning site (fore-reef) with giant barrel sponge (*Xestospongia muta*), *Orbicella* sp. coral patch in sand and *Acropora palmata* coral on fore-reef, green turtle (*Chelonia mydas*) over seagrass (*Thalassia testudinum*) and caesar grunts (*Haemulon carbonarium*) with giant barrel sponge (*Xestospongia muta*) on fore-reef. All photos © A. Tewfik.

3. Fisheries Overview

3.1 FAO Data, Reconstructions and Assessments

Caribbean spiny lobster, queen conch and marine fish (nei, not elsewhere indicated) form the bulk (73%) of the exported wild caught products from Belize (within FAO Area 31) in the 21st century (FAO 2023). Lobster, conch and fish represent 47%, 19% and 7% respectively with increasing exports of lobster and conch (meat only) while marine fish landings have declined steadily beginning in 2013 (Fig. 1) (UNCTAD 2022, FAO 2023). Recent work indicates that the marine fish nei category often includes a previously hidden source of fishing mortality for many threatened species of sharks and rays (MacNeil et al. 2025). Lobster and conch data represent single species that may be exported in various forms: whole lobster, tails, head meat, boiled or frozen, and conch frozen at various processing levels (i.e., progressive removal of soft tissue to white fillets).

Significant levels of other invertebrate catch are reported for Belize including various shrimp, stone crab (*Menippe mercenaria*) and sea cucumber species. Wild caught shrimp first appeared in 1967 and represented up to 15% of exports during the early 21st century but ended in 2010 with the banning of trawlers (Greers et al. 2020). Stone crab landings first appear in 1981, with a peak of 210 MT in 1997, and ending in 2016. A sea cucumber fishery had existed for ~20 years through the local Asian market and with trade through Guatemala (Rogers et al. 2018). The broader international export of sea cucumbers began with the establishment of regulations in 2009 (Rogers et al. 2018). This export fishery was quite short lived, with rapid expansion beginning with exports of 24 MT in 2010, peaking at 587 MT (2013) and ending in 2017 with 18 MT. This classic boom to bust scenario is blamed on inadequate enforcement and research as well as poor education of fishers on the socioeconomic dangers of overfishing (Rogers et al. 2018).

Data (1976 – 2005) from the two largest co-ops supplied by the Belize Fisheries Department indicates highly variable finfish exports (≈ 225 to 450 MT) with the majority being grouper and snapper, throughout the late 1970s, 80s and early 1990s, with a 20-fold decrease from highs of ≈ 450 MT in 1976 to ≈ 23 MT by 2005, (Heyman and Wade 2007). Additional reports for the major commodities of conch and lobster verify the approximate scale and proportion of exported products reported by FAO (Gibson et al. 2004). It is noteworthy that FAO landings data from 2017 – 2022 are approximated, according to the status code in the FAO capture production dataset, as no data were submitted by the Government of Belize. What is not clear for the FAO marine fish nei category is the tremendous diversity of bony fish, and possibly shark (MacNeil et al. 2025), taxa (90 species, 34 families), size (Lmax = 30 to 250 cm), trophic level (2.2 to 4.5) and food web interactions (Fig. 6) that this catch-all category represents (Tewfik et al. 2022). Several pelagic groups of fish (billfish, tunas, blue shark (*Prionace glauca*), shortfin mako shark (*Isurus oxyrinchus*), dolphinfish (*Coryphaena hippurus*)) are reported by FAO separately for Belize. These generally do not represent reef complex species caught within national waters (i.e., within the EEZ) by local artisanal fishers and comprise catch by Belize flagged vessels in other jurisdictions within Area 31. As an example, only three shortfin mako have been observed across numerous shark studies in Belize's barrier reef complex (see section 5.6). In contrast, FAO indicated 23 and 28 metric tonnes of shortfin mako were landed in 2009 and 2010, respectively (FAO 2023). Catch of sharks, rays, and skates (nei) was first reported by FAO in 1997 (1 MT), peaking in 2005 (10 MT) and ending in 2011 (6 MT), and would likely include Belize reef complex species caught by local special licenced fishers. However, shark exports have been reported as taking place for decades, beginning in the 1960s (Graham 2007), with registered exports through the Fisheries department documented in the mid-90s (Gibson et al. 2004) and do not seem to appear in FAO statistics, although they could be part of the marine fishes nei category (MacNeil et al. 2025).

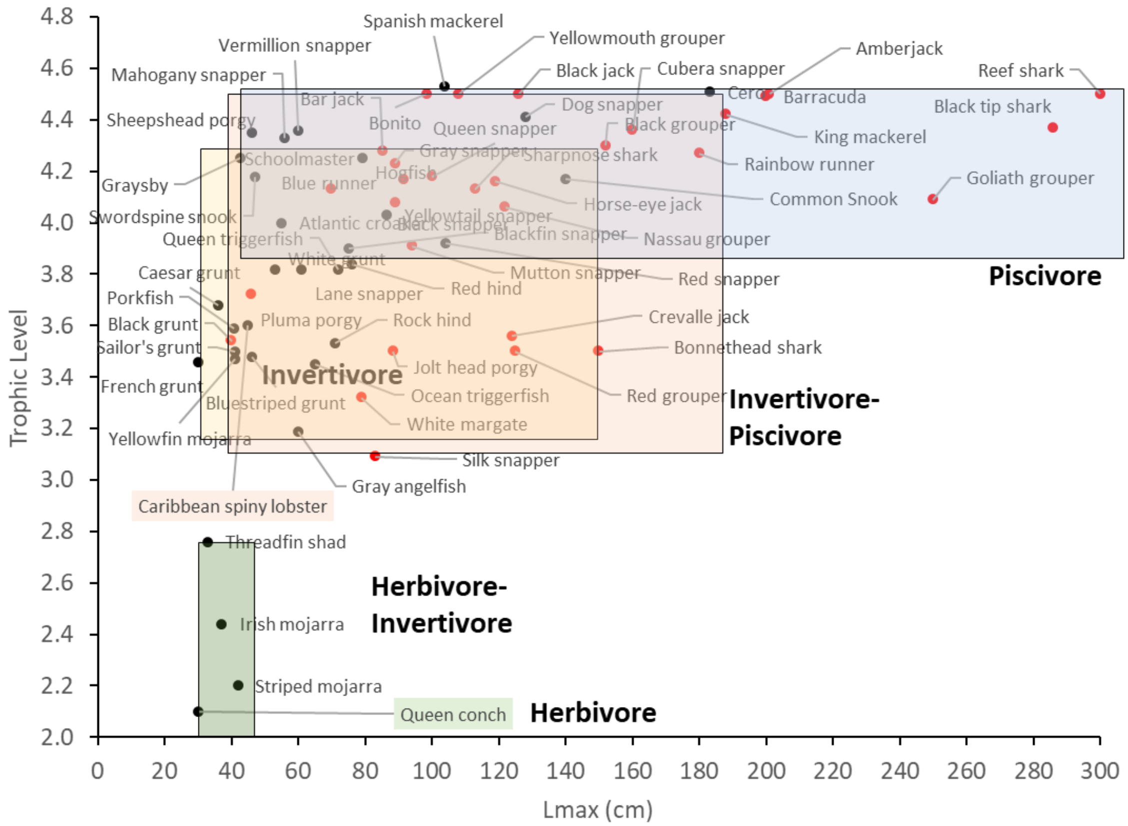


Figure 5: Biplot of maximum length (Lmax – total length, shell length, or carapace length) and trophic level for 56 species of bony fish (Tewfik et al. 2022), 4 species of shark (Quinlan et al. 2021), queen conch (Tewfik et al. 2019) and Caribbean spiny lobster (Tewfik et al. 2020) reported landed in Belize. Colored trophic group polygons (green – herbivore-invertivore, yellow – invertivore, red – invertivore-piscivore, blue – piscivore) are anchored by extremes in size and trophic level of species in each trophic group and depict the overlapping interactions of species within reef complex habitats. Fish with red points are considered overfished using the ratio of mean length caught/length of maturity (Lx/Lm) (See Babcock et al. 2018 and Figure 8 below). See text for overfished status of other non-bony fish species.

While FAO capture production and export data can provide a useful understanding of the relative scale and importance of extracted commodities, it is based on self-reported data, which are often incomplete for most countries (Zeller et al. 2011). In addition, the estimates provided by FAO in the absence of country reported data (e.g., Belize data from 2017 – 2022 for all wild caught species) may not accurately reflect changes in catch composition from local fishing grounds. The incomplete collation of complex, multi-species wild capture fisheries landings at local, national and global scales have inspired the development of a number of data reconstruction approaches and associated assessments using historical data from multiple sources (Pauly and Zeller 2016, Ulman et al. 2016, McClenachan et al. 2024). Belize has undergone such catch reconstructions with the estimate of total catches over 3.5 times larger than values reported in the FAO database, averaging around 6,000 mt·year-1 live weight in total since 2000 (Zeller et al. 2011). This major discrepancy was mainly due to: (1) catches focused on those sold through the fishing cooperatives and exported; (2) unreported catches of sharks; (3) under-reported subsistence catches; and (4) unreported uses in domestic markets and the local tourism sector –all of which contribute to local livelihoods. IUU fishing by vessels from neighbouring countries was also a major concern for Belize (Zeller et al. 2011, Phillips et al. 2025). Reconstruction techniques improve understanding of landing patterns and scale, and can potentially be used to provide more accurate estimates of status of harvested populations to inform sustainable management.

The most recent fisheries reconstructions and associated species stock assessments for Belize were conducted by the Sea Around Us Project (SAUP), Institute for Oceans and Fisheries, University of British Columbia (Palomares et al. 2023). Within the EEZ of Belize (1950 - 2018) fisheries production is largely exported (skewed to queen conch due to reporting as live weight with shell, Fig 1) and dominated by artisanal (67%) and subsistence (11%) activities with industrial and recreational fisheries consisting of 11% each (Palomares et al. 2023). Catch maximum sustainable yield (CMSY++) (Froese et al. 2017) models were used to determine biomass trajectories for 19 commonly fished species including two invertebrates (queen conch, Caribbean spiny lobster), three groupers (Nassau, black, goliath), seven snappers (e.g., mutton (*Lutjanus analis*), yellowtail (*Ocyurus chrysurus*)), two jacks (crevalle (*Caranx hippos*), horse-eye (*Caranx latus*)), great barracuda (*Sphyraena barracuda*), king mackerel (*Scomberomorus cavalla*), and common snook (*Centropomus undecimalis*). Abundance maximum sustainable yield (AMSY) models (Froese et al. 2020) were used for two elasmobranchs (scalloped hammerhead (*Sphyrna lewini*), Caribbean reef shark (Carcharhinus perezi)) where no viable catch data were available. Final positions of trajectories are shown on a Kobe plot (FAO 2007) with the vast majority (84%) of species assessed as being unsustainably overfished (Palomares et al. 2023) (Fig. 7). Such a result is unsurprising as 51% of landed individuals, regardless of taxonomic family, landing site or gear, are landed immature (Tewfik et al. 2022). The SAUP results included the two most important exported species (queen conch and Caribbean spiny lobster) and a number of coral reef dependent snappers and groupers (e.g., critically endangered Nassau grouper) that have provided the mainstay of bony fish harvest over the last 100 years (Tewfik et al. 2019, Tewfik et al. 2020, Tewfik et al. 2022, Phillips et al. 2025). Only the horse-eye jack was determined to be in a healthy and sustainably fished state (Fig. 7).

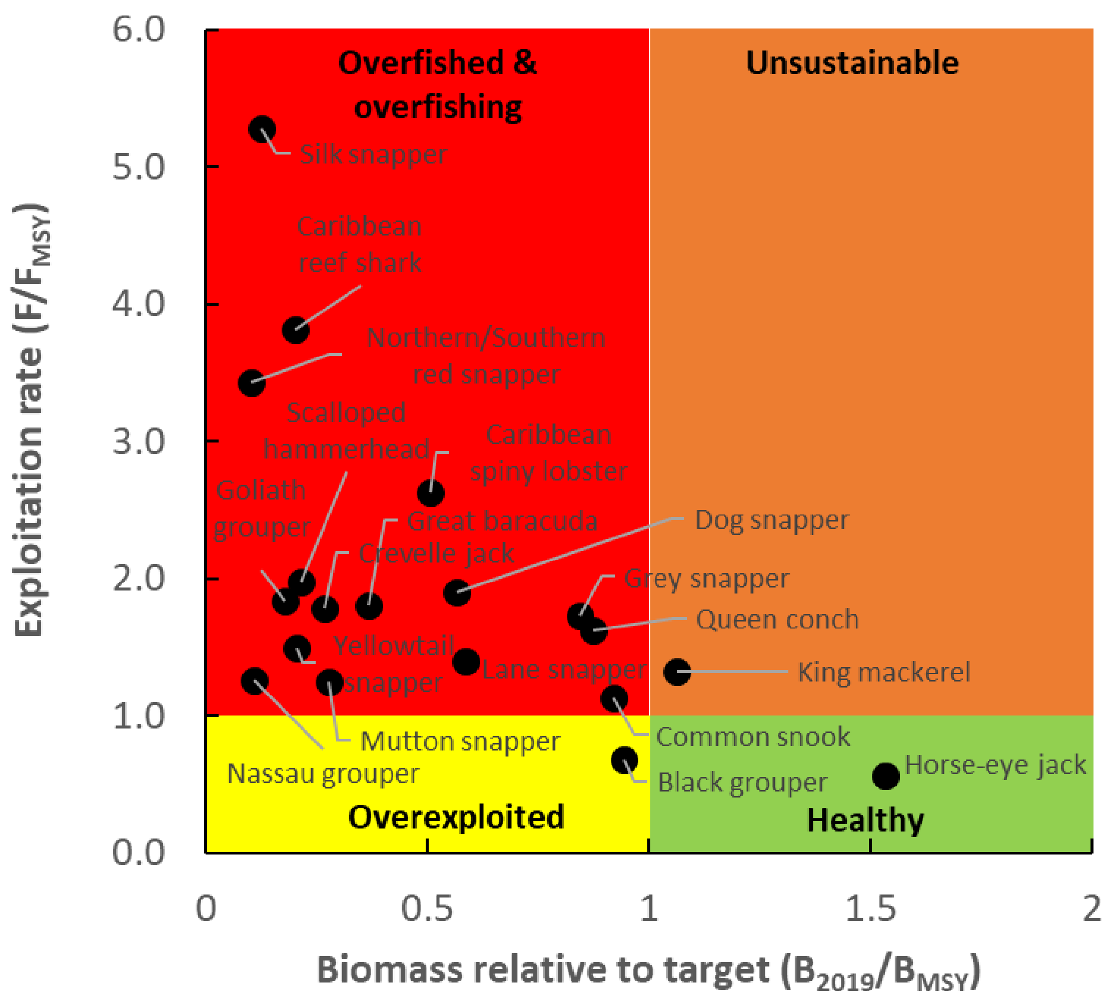


Figure 6: Kobe plot for 19 commonly fished species within Belize's reef complex habitats. Species positions indicate final point of reconstructed catch (abundance for two elasmobranchs with no viable catch data) trajectories in 2019. Northern and Southern red snappers were combined in a single analysis given similarity of life histories. Redrawn from Palomares et al. 2023.

3.2 On-the-Ground Landings Data

Local fishing activity may be best tracked by on-the-ground studies, which regularly and directly interact with catch and fishers and have the added benefits of collecting data on individual fish and invertebrates (e.g., length, biomass). Such data collection programs may also inform global databases and catch reconstruction efforts. These direct interactions also allow data collectors, scientists and resource managers to collect local knowledge and hear concerns as well as convey best practices directly to resource stakeholders (King 1997, Alves et al. 2022, García-Téllez et al. 2022, McClenachan et al. 2024). The catch and perceptions of local fishers, most with many years of experience (70% > 10 years, 20% > 30 years), were examined in southern Belize (Heyman and Graham 2000). They found that individual target species were said to be declining in size (67%) and were lower in number (70%) with lobster, conch, cero mackerel (*Scomberomorus regalis*), lane snapper (*Lutjanus synagris*) and yellowtail snapper (*Ocyurus chrysurus*) topping the list of declining populations. The main reasons for these declines were given as overfishing, smuggling and cross border fishing (Heyman and Graham 2000).

More recent catch studies are available for Glover's Reef Marine Reserve (Tewfik 2018), Southwater Caye Marine Reserve (Tewfik et al. 2018), and at major settlement landing sites and markets along the coast of Belize (Tewfik et al. 2022) (Fig. 3). Together these data represent 19,822 individually identified and measured fish collected between 2017 and 2020. Ten families represented 98.4% of all fish landings with the remaining 1.6% representing 23 families with just 1 - 64 individuals in each (Fig. 8a). Snappers (Lutjanidae) were the most caught family representing 46%, followed by jacks (Carangidae), mojarras (Gerridae), grunts (Haemulidae), and mackerels (Scombridae) (Fig. 8a). The historically dominant groupers (Serranidae) represented just 2.6%. Snappers are also the most caught family (45 – 56%) in all TURFs (no data available from area 7) with the exception of Area 1 where mojarras (34%) edged out snappers (31%). Traditional fisheries for mojarras in Area 1 (concentrated in turbid Corozal Bay) using beach weirs and gillnets are largely responsible for this exception. Invertivore-piscivores (mixed diet consumers) (Babcock et al. 2018) are the largest represented trophic group for these landings followed by invertivores and piscivores with very few herbivore-invertivores (Fig. 8b). When examining the relationship between an index of overfishing (Lx/Lm, mean caught length / size at maturity) and Lmax (maximum recorded size for the species) (Babcock et al. 2018) the larger species were generally more overfished (Lx/Lm < 1.0) (Fig. 9). Across all taxa, 51% of landed individuals were immature, with larger piscivores and invertivore-piscivores across many families having few mature fish appearing in the landings. This foreshadows problems for future reproduction, which is critical for recruitment and sustainable fisheries (Tewfik et al. 2022).

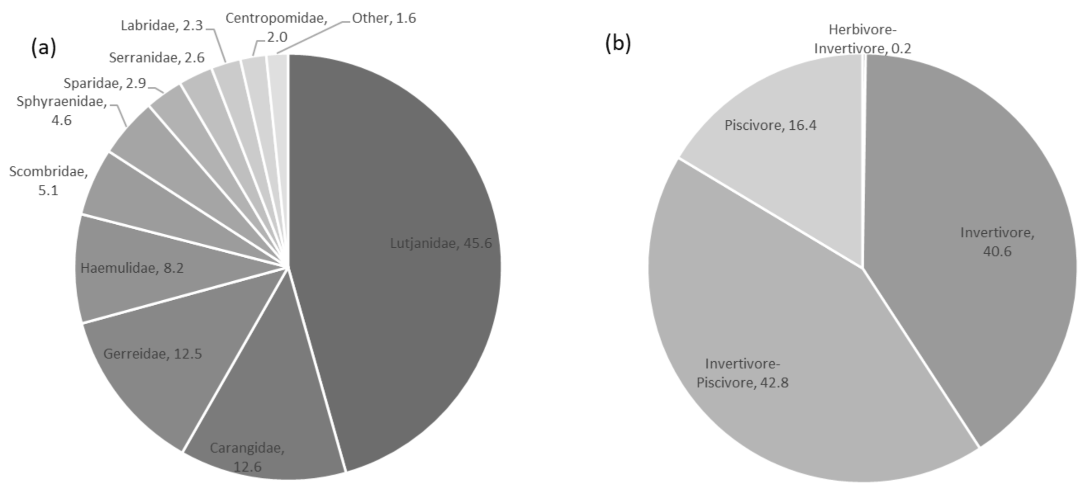


Figure 7: Distribution of individuals within (a) families and (b) trophic groups from catch landings data collected in Belize between 2016 and 2020. 7 of 8 TURFS (no data for Area 7), n = 19822 (Tewfik 2018 a, b; Tewfik et al. 2022).

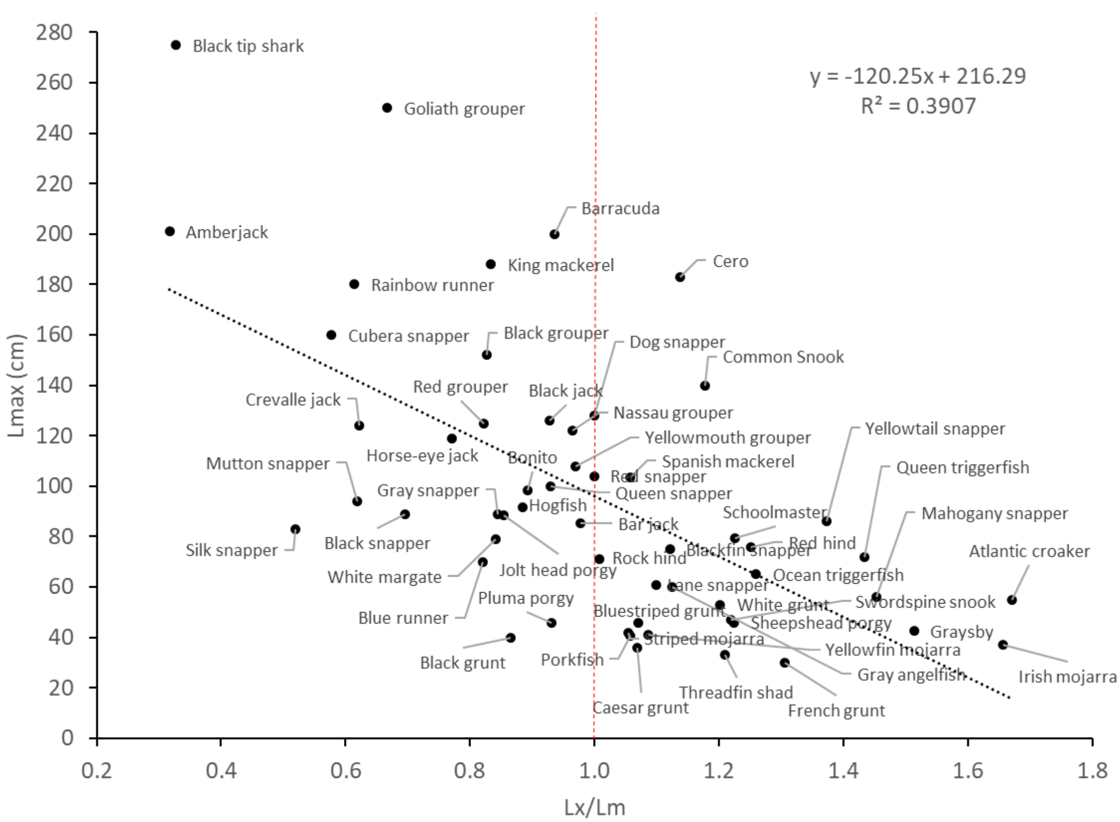


Figure 8: Biplot of an index of overfishing ((Lx/Lm, mean caught length / size at maturity) (Babcock et al. 2018) and Lmax (maximum size for the species) for 56 bony fish and one shark landed in Belize. Red line indicates threshold for overfished status (> 1.0). Detailed size frequency and gear data for barracuda, crevalle jack and yellowfin mojarra, goliath, black and red hind groupers, cubera, mutton and lane snapper may be found in figure. 13.

When comparing artisanal fisher landings data for shared bony fish taxa from the late 20th century (Heyman and Graham 2000) to more recent data (Tewfik et al. 2022) (Table 1), three of four trophic groups declined in representation in the catch. Only invertivore species of small size and low trophic level increased in representation over time (Fig. 10a). The top 5 families all show decreases in representation in the catch ranging from 17 to 44 percent (Fig 10b). Across the 33 individual species compared, 20 showed some decline in percentage of catch over the two-decade timeframe between studies including yellowtail snapper (20 to 6%), mutton snapper (*Lutjanus analis*, 18 to 5%) and cero mackerel (*Scomberomorus regalis*, 8.5 to 1.3%) (Table 1). The resulting decline in the Shannon diversity index ($H' = \Sigma(pi * \ln(pi))$, where pi = proportion of individuals in a species) from the early (2.643) to the later period (2.267) is 14.2% for the 33 shared species.

Table 1: Shared species between Belize landings data reported in the late 20th century (Heyman and Graham 2000) and more recent data (Tewfik et al. 2022). Lmax = maximum total length (cm reported for the species), TL = trophic level and trophic group (Tewfik et al. 2022).

Species	Family	Lmax	TL	Trophic group	% (Heyman & Graham)	% (Tewfik)	change
Selene vomer	Carangidae	48	3.9	Invertivore	0.77	0.00	-0.77
Caranx hippos	Carangidae	124	3.6	Invertivore-Piscivore	7.28	4.15	-3.13
Caranx latus	Carangidae	119	4.2	Invertivore-Piscivore	0.77	3.17	2.40
Alectis ciliaris	Carangidae	150	3.8	Invertivore-Piscivore	0.77	0.00	-0.77
Trachinotus goodei	Carangidae	50	4.3	Invertivore-Piscivore	0.06	0.01	-0.06
Centropomus undecimalis	Centropomidae	140	4.2	Invertivore-Piscivore	1.02	1.95	0.93
Chaetodipterus faber	Ephippidae	91	4.5	Invertivore	0.77	0.01	-0.77
Gerres cinereus	Gerreidae	41	3.5	Invertivore	0.77	6.80	6.03
Haemulon sciurus	Haemulidae	46	3.5	Invertivore	3.24	1.42	-1.82
Holocentrus marianus	Holocentridae	18	3.6	Invertivore	0.77	0.00	-0.77
Lachnolaimus maximus	Labridae	92	4.2	Invertivore	0.77	1.95	1.18
Lutjanus synagris	Lutjanidae	61	3.8	Invertivore	9.80	14.73	4.93
Ocyurus chrysurus	Lutjanidae	86	4.4	Invertivore-Piscivore	20.01	5.80	-14.21
Lutjanus analis	Lutjanidae	94	3.9	Invertivore-Piscivore	18.02	5.13	-12.89
Lutjanus cyanopterus	Lutjanidae	160	4.4	Invertivore-Piscivore	1.07	0.65	-0.42
Lutjanus jocu	Lutjanidae	128	4.4	Invertivore-Piscivore	1.07	2.39	1.32
Lutjanus griseus	Lutjanidae	89	4.2	Invertivore-Piscivore	1.07	4.04	2.97
Lutjanus campechanus	Lutjanidae	100	3.9	Invertivore-Piscivore	1.07	0.48	-0.59
Lutjanus apodus	Lutjanidae	79	4.3	Invertivore-Piscivore	1.07	2.21	1.14
Mugil cephalus	Mugilidae	100	2.3	Herbivore-Invertivore	0.77	0.16	-0.61
Mulloidichthys martinicus	Mullidae	45	3.4	Invertivore	0.77	0.00	-0.77
Rachycentron canadum	Rachycentridae	214	4.0	Piscivore	0.77	0.06	-0.71
Pogonias cromis	Sciaenidae	170	3.4	Invertivore-Piscivore	0.77	0.02	-0.75
Scomberomorus regalis	Scombridae	183	4.5	Piscivore	8.48	1.29	-7.19
Scomberomorus maculatus	Scombridae	104	4.5	Piscivore	0.57	2.69	2.12
Scomberomorus cavalla	Scombridae	188	4.4	Piscivore	0.11	1.14	1.03
Epinephelus itajara	Serranidae	250	4.1	Invertivore-Piscivore	0.67	0.17	-0.50
Epinephelus guttatus	Serranidae	76	3.8	Invertivore-Piscivore	0.47	0.90	0.43
Epinephelus striatus	Serranidae	122	4.1	Invertivore-Piscivore	0.30	0.10	-0.20
Mycteroperca bonaci	Serranidae	152	4.3	Piscivore	0.20	0.20	0.00
Archosargus rhomboidalis	Sparidae	8	3.0	Herbivore-Invertivore	0.77	0.00	-0.77
Calamus calamus	Sparidae	56	3.5	Invertivore	0.77	0.03	-0.74
Sphyraena barracuda	Sphyraenidae	200	4.5	Piscivore	4.34	4.39	0.05

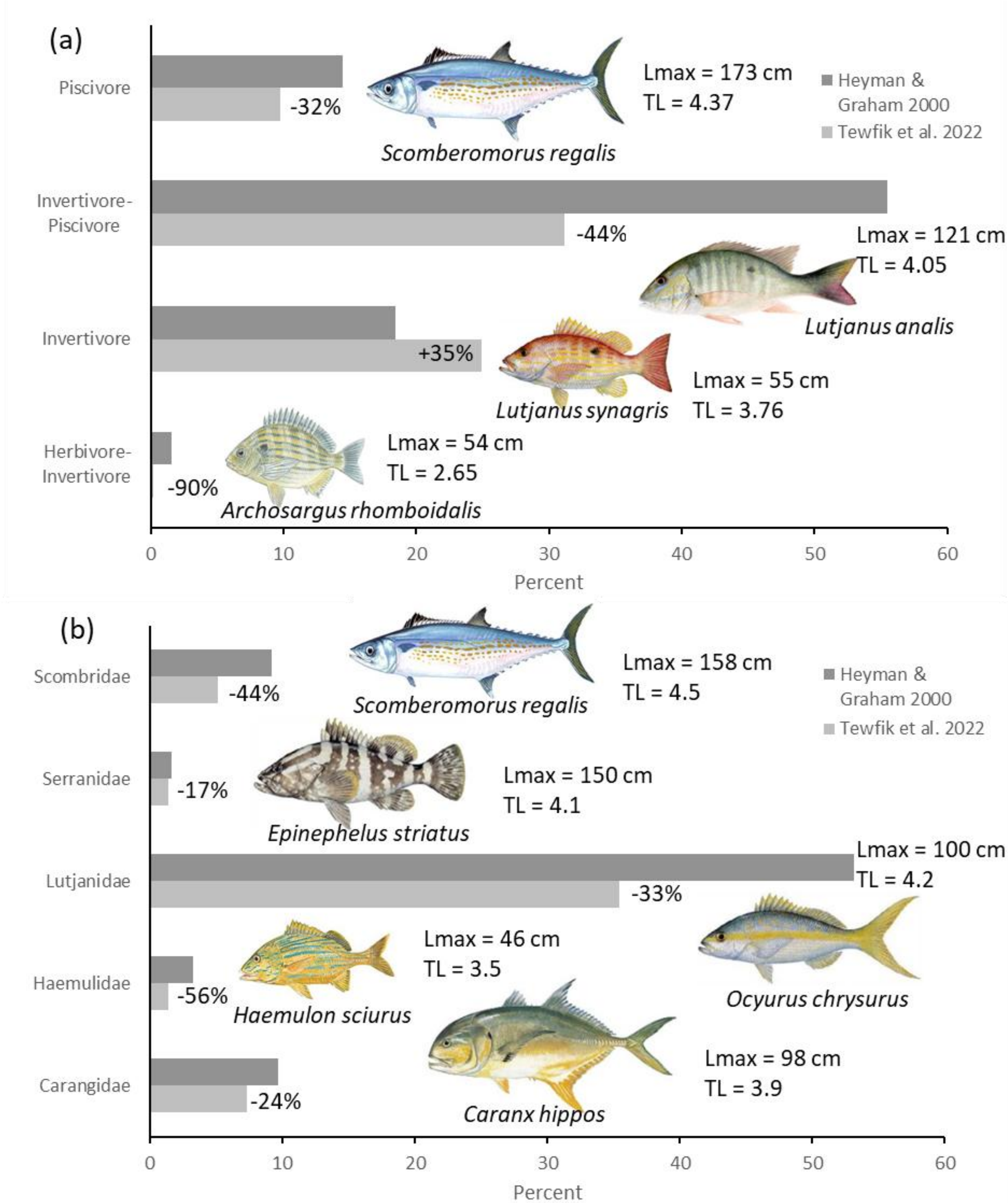


Figure 9: Changes in the representation of major bony fish (a) trophic groups and (b) families over 20 years based on early (Heyman and Graham 2000) and later accounts (Tewfik et al. 2022) of landings across Belize. Only shared bony fish taxa (16 families, 33 species) between the two studies were compared. Percent change is indicated for each group pair and a single species model (not scaled) represents each. Mean maximum length (Lmax) and mean trophic level (TL) are given for all species in each trophic group and family (Table 1). Decline in the Shannon diversity index (H'= Σ pi * ln(pi)), where pi = proportion of individuals in a species) from the early (2.643) to the later period (2.267) is 14.2% for the 33 shared species.

The Healthy Reef Index (HRI) reports describe general fishery-independent trends from visual surveys over time using indices for herbivorous (i.e., parrotfish and surgeonfish) and commercial (snappers and groupers) species calculated as average fish biomass across 110 sites in Belize (McField et al. 2024a). Herbivorous fish biomass increased to levels considered good (2740 g/100 m2) by 2018 with lower numbers subsequently despite the complete ban on parrotfish harvest since 2009 (Fig. 11a). The encouraging level of herbivorous fish biomass observed in some years is likely a combination of legal protection and changing trophic dynamics (Mumby et al. 2012), including reduced predation due to overfishing and enriched algal food supplies due to land-based nutrient inputs (Lapointe et al. 2021). Commercial fish were at fair levels (800 g/100 m^2) or below across the entire time series, reaching critical (< 390 g/m^2) levels in 2021, possibly due to increased subsistence fishing during the COVID-19 pandemic, with some subsequent recovery in 2023 (Fig 11b).

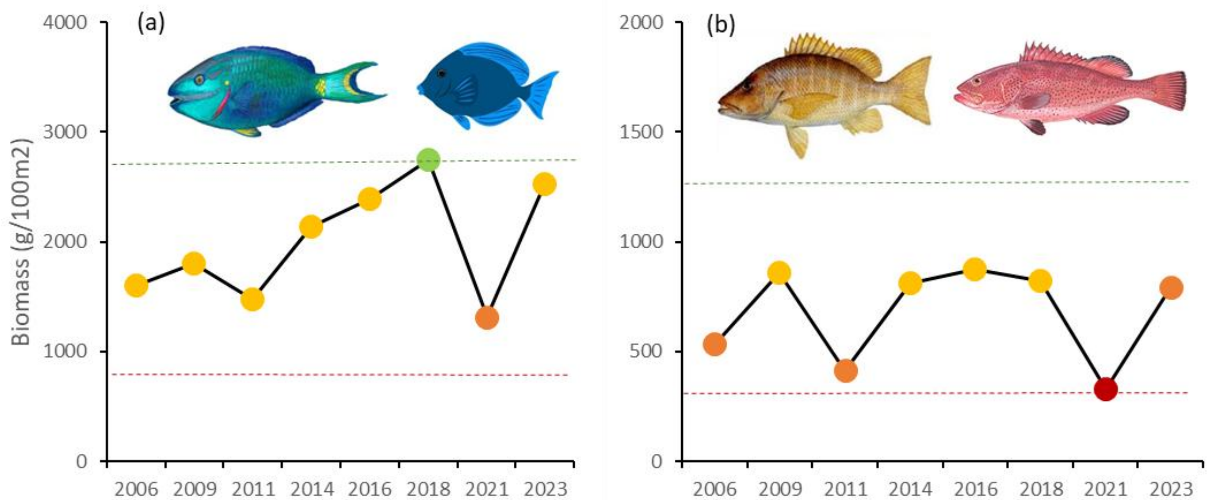


Figure 10: Belize (a) herbivorous (parrotfish and surgeonfish) and (b) commercial (snapper and grouper) fish indices (biomass – g/100 m2). Data and criteria from McField et al. 2024, supplements. Symbol colors: orange (poor), yellow (fair) and green (good), red lines indicate threshold for critical states (herbs < 990, commercial < 390) and green lines indicate thresholds for a good state (herbivores > 2740, commercial > 1210).

3.3 Fishing Gear Impacts

Local data collection and discussions with fishers may also highlight details pertaining to the use and impact of fishing gear across the wide range of fish species (> 90) harvested in Belize. Details are available on the use of five gear categories (line, gillnet, other net, spear, trap) for 17,759 individuals across 35 families, 89% of individuals within the top 7 families (Fig. 12), collected between 2017 and 2020 (Tewfik et al. 2022). Most fish were caught using line (& hooks), followed by gillnet, trap, other net, and spear (Fig. 12). It should be noted that gillnets were legal during the study period and were not banned in Belize until Nov. 2020 (Greers et al. 2020). Within the major finfish families described, line was the most used gear (50.9 – 88.0%) except in the case of the mojarras (Fig. 12), which were caught mostly with gillnets (42%), followed by traps (27%) (Fig. 12c). Of the other 6 most caught families, jacks and mackerels were the most impacted by gillnets (30.3% and 21.4%) (Fig. 12d, f), with snappers and groupers being caught by gillnets at lower rates (16.5% and 3.2 %) (Fig. 12b, h). Prior to the ban of gillnets, most fishers targeting sharks primarily used monofilament gillnets and bottom longlines (Greers et al. 2020).

Gillnets were banned to facilitate the live release of immature fish that might otherwise expire in over-night or multiday gillnet sets. However, given that there are no size limits on any commercial fish species except for the Nassau grouper (UNCTAD 2022), immature fish caught in gillnets are generally retained (Tewfik et al. 2022), unless damaged by secondary predation. Smaller, immature sizes within species may be used for subsistence, private trade or bait with larger, higher valued individuals entering the domestic commercial market or used for export. Nevertheless, preventing excessive harvest of immature individuals would likely make the fishery more sustainable by allowing more fish to grow larger and potentially reproduce before being caught (Froese 2004).

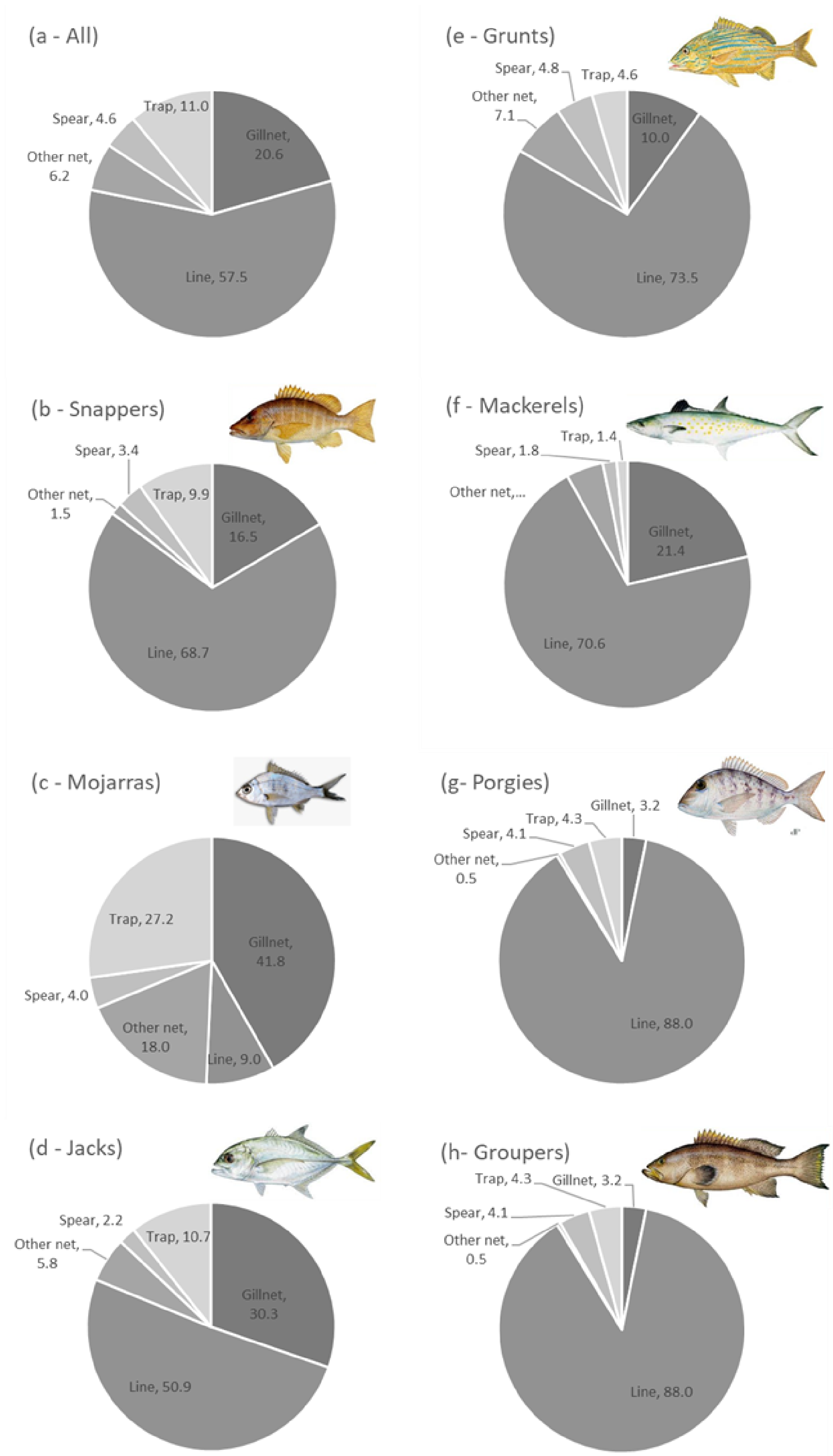
Size frequencies of catches by gear of small, medium and large species across some of the most landed families were examined (Fig. 13) (Tewfik et al. 2022). The great barracuda (*Sphyraena barracuda*) is amongst the longest bony reef fish (Lmax = 200 cm) in the Caribbean and regularly caught in Belize (Froese and Pauly 2021, Tewfik et al. 2022). Barracuda were ranked as the fourth most caught species by fishers in the northern region of Corozal (Greers et al. 2020) and were also reported to be the most frequently caught species at Glover's Reef in 2007 (Greers et al. 2020), declining to fourth most caught in 2016 due to ongoing exploitation (Tewfik 2018). This single species (family Sphyraenidae) accounted for almost 5% of the total finfish landings recorded nationally (Fig. 8a) (Tewfik et al. 2022). All five gear types were used to capture barracuda with line (55%), gillnet (20%) and traps (18%) accounting for most of the landings (Fig. 13). The opportunistic targeting of prey fish by barracuda struggling in various gear (i.e., nets) may account for all gears catching barracuda. A large percentage of barracuda landings (61%) were under the Lm (71 cm), with the largest size (132 cm) encountered being well under the known Lmax (200 cm) for this species (Fig. 13) (Froese and Pauly 2021). While barracuda are rarely considered amongst threatened species in the Caribbean reef complex, they are large, high-trophic level predators (Bond et al. 2018), frequently targeted in Belize especially at immature sizes, so they probably require management attention. Similarly, another large reef complex associate, the common snook (*Centropomus undecimalis*, Centropomidae) (Lmax = 140 cm), constituted 2% of the catch with 62% of the catch being mature illustrating that large fish, based on Lmax, are often caught (Fig. 9).

The crevalle jack (*Caranx hippos*) is a large reef-associated coastal pelagic travelling in small groups and a known target of Belizean fishers (Heyman and Graham 2000). Landings are largely achieved using line (48%) and gillnets (42%) (Fig. 13), and over 92% of landed crevalle jacks were below the Lm (64 cm) with the largest individual (90 cm) being only 72% of Lmax (124 cm) (Froese and Pauly 2021). A number of other coastal reef associated jacks and mackerels were frequently caught by gillnets including the blue runner (*Caranx crysos*), horse-eye jack (*Caranx. latus*), king (*Scomberomorus cavalla*), cero (*Scomberomorus regalis*) and Spanish mackerel (*Scomberomorus maculatus*) (Greers et al. 2020).

The yellowfin mojarra (*Gerres cinereus*, n = 1258, Lmax = 41 cm) and the closely related striped mojarra (*Eugerres plumieri*, n = 698, Lmax = 42) are amongst the smallest exploited species in Belize and together form the second largest family (Gerridae) caught nationally and the top (33.6%) family exploited in TURF area 1 (Tewfik et al. 2022). Even for the small, fast-growing and early maturing (Lm = 21 cm) yellowfin mojarra, approximately 15% were landed immature (Fig. 13). Almost 56% of these fish were landed using gillnets followed by traps (30%), including beach weirs, and line (13%) (Fig. 13). It is clear that the banning of gillnets will have a significant effect on the harvest of mojarras, especially in northern Belize TURF Area 1 where they are most targeted.

For groupers, the largest species, goliath, has almost all individuals (97%) landed below the size of maturity (Lm = 121 cm). The large groupers and snappers are mainly taken by hook and line, followed by spear, and none are documented as taken using gillnets, a recently banned gear, in the legal fishery (Tewfik et al. 2022) (Fig. 13). Black grouper has a reduced but still high percentage (74%) of individuals landed under Lm (63 cm), again without the use of gillnets, and red hind has a smaller proportion of individuals (25%) landed under Lm (25 cm), and with a small percentage (10.3%) caught using gillnets (Fig. 13). A similar pattern appears for snappers with the largest species, cubera, being landed largely (95%) under the Lm (54 cm) but with gillnets capturing almost 25% (Fig. 13). The medium sized and heavily utilized mutton snapper (Graham et al. 2008) had 88% of individuals landed under the Lm (50 cm), with only 7% landed using gillnets (Fig. 13). The small lane snapper, the most caught bony fish species across Belize (Tewfik et al. 2022), was landed under Lm (22 cm) 16% of the time, but only 9.3% of the landings were made using gillnets (Fig. 13).

Across finfish species and gears, a distinct pattern emerges with the largest species (barracuda, goliath, cubera) subject to the largest immature fractions caught and smaller species seeing significantly lower proportions caught below the size of maturity (Fig. 9). While gillnets contribute to the retention of immature individuals across families, their ban alone cannot reduce the overall high level of immature landings, especially of large species, and associated impacts on future recruitment and productivity. The addition of minimum legal sizes across the diversity of commercially caught species might allow rebuilding of declining populations of the large (goliath, cubera) and medium (black grouper, mutton snapper) sized species, and protect smaller species that still have productive populations (e.g., lane snapper, red hind, *Epinephelus guttatus*) used for domestic consumption and increasingly for export as large and medium sized species become rare.



(a - All)
Trap, 11.0
Spear, 4.6
Other net, 6.2
Gillnet, 20.6
Line, 57.5
(e - Grunts)
Spear, 4.8
Trap, 4.6
Other net, 7.1
Gillnet, 10.0
Line, 73.5
(b - Snappers)
Spear, 3.4
Other net, 1.5
Trap, 9.9
Gillnet, 16.5
Line, 68.7
(f - Mackerels)
Trap, 1.4
Spear, 1.8
Other net,...
Gillnet, 21.4
Line, 70.6
(c - Mojarras)
Trap, 27.2
Gillnet, 41.8
Spear, 4.0
Other net, 18.0
Line, 9.0
(g- Porgies)
Gillnet, 3.2
Trap, 4.3
Spear, 4.1
Other net, 0.5
Line, 88.0
(d - Jacks)
Spear, 2.2
Other net, 5.8
Trap, 10.7
Gillnet, 30.3
Line, 50.9
(h- Groupers)
Trap, 4.3
Gillnet, 3.2
Spear, 4.1
Other net, 0.5
Line, 88.0

Figure 11: Proportion of landings by gear (line, gillnet, other net, spear, trap) in Belize. (a) All fish (n = 17759), (b) snappers (Lutjanidae) (n = 8031), (c) mojarras (Gerreidae) (n = 2426), jacks (Carangidae) (n = 2139), grunts (Haemulidae) (n = 1425), mackerels (Scombridae) (n = 931), porgies (Sparidae) (n = 443), and groupers (Serranidae) (n = 424) (Tewfik et al. 2022).

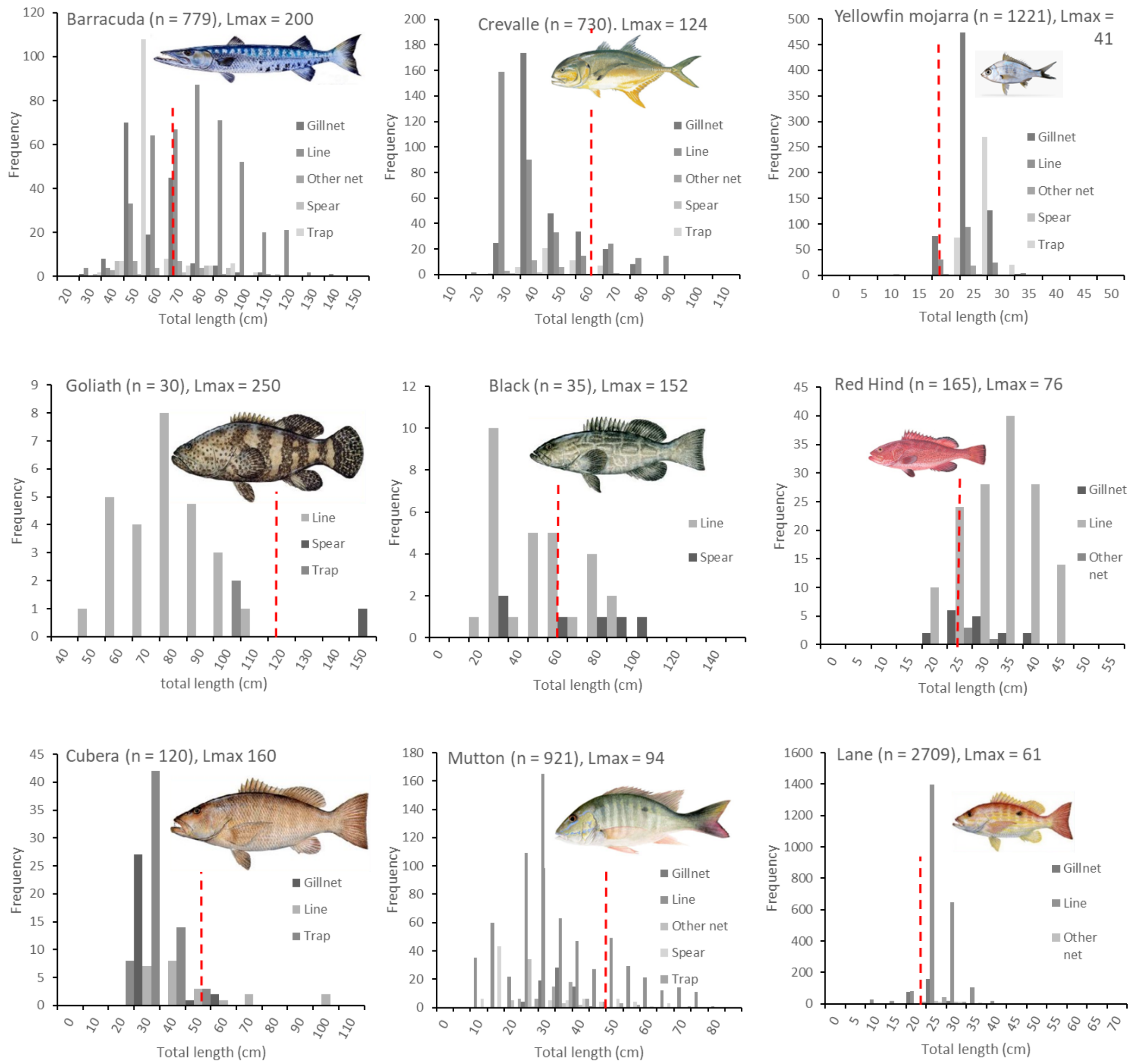


Figure 12: Commercial landings of common species in Belize spanning overall size range by gear type between 2017 and 2020 prior to gillnet ban (Tewfik et al. 2022). Top row: large (great barracuda (*Sphyraena barracuda*), Lmax = 200 cm), medium (crevalle jack (*Caranx hippos*), Lmax = 64 cm) and small (yellowfin mojarra (*Gerres cinereus*), Lmax = 21 cm) representing Sphyraenidae, Carangidae and Gerridae. Middle row: large (Atlantic goliath grouper (*Epinephelus itajara*), Lmax = 250 cm); medium (black grouper (*Mycteroperca bonaci*), Lmax = 152 cm); and small (red hind (*Epinephelus guttatus*), Lmax = 76 cm) (Serranidae). Bottom row: large (cubera snapper (*Lutjanus cyanopterus*), Lmax = 160 cm), medium (mutton snapper (*Lutjanus analis*), Lmax = 94 cm) and small (lane snapper (*Lutjanus synagris*), Lmax = 61 cm) (Lutjanidae). Dashed red line is the length of maturity. Fish models are not exactly scaled to each other.

4. Principal Fisheries

4.1 Queen conch (*Aliger gigas*)

The queen conch has been targeted by fishers in the area of Belize for over 1500 years as evidenced by conch shells "knocked" using primitive tools in Mayan food refuse middens, with the modern commercialization of the species beginning in the 1930s (Craig 1966, McKillop 1984). The harvesting of conch has developed into one of the most important and iconic fisheries within the region and Belize, supporting subsistence food gathering and critical livelihoods related to local markets, tourism, and significant exports largely to the United States, Canada and European Union countries and territories (Fig. 1) (Tewfik 1997, Theile 2001, Prada et al. 2017). Despite the large body of science available on this CITES Appendix II listed species, several unique aspects of conch growth and reproductive ecology as well as the relative ease of exploitation by breath-hold (the only legal technique in Belize), surface-supplied and SCUBA divers, have proved challenging to local management across the region (Prada et al. 2017, Boman et al. 2018, Stoner and Appeldoorn 2021).

Conch have a two-phase pattern of shell growth: (1) juveniles maximizing shell length (SL) to avoid a host of predators (Ray and Stoner 1995, Tewfik 1997), and (2) the cessation of shell length growth and commencement of shell thickening by sub-adults at the onset of maturation to produce a thick, flared shell lip - the most reliable indicator of sexual maturity (Alcolado 1976, Appeldoorn 1988, Buckland 1989). Given the same shell length, conch with a flared and thick shell lip have more soft tissue than unlipped conch (juveniles) (Stoner et al. 2012b). Also, since growth in SL ceases upon lip formation, larger adults have more gonad mass than small adults (Stoner et al. 2012b). These have significant implications for commercial production (i.e., meat yield), population fecundity and recruitment success (Kough et al. 2017, Stoner et al. 2018, Tewfik et al. 2019). A number of studies have found that shell lip thickness (LT) of 10 mm or greater best indicates sexually mature conch (Avila-Poveda and Baqueiro-Cárdenas 2006, Stoner et al. 2018) including two studies in Belize conducted in different marine reserves using different approaches (Foley and Takahashi 2017, Tewfik et al. 2019). These latter two studies calculated a threshold for the size at 50% maturity of 10 – 16 mm thick shell lip and an associated 192 - 199 g "market clean" (i.e. semi-fillet) meat mass. This is in sharp contrast to size regulations for the harvest of queen conch in Belize, which permit a minimum shell length of 178 mm and directly associated "market clean" meat mass of 85 g with no use of lip thickness (Anonymous 2020). These regulations are holdovers from the first species catch limits enacted in 1978, which were meant to be temporary as more information was gained about the species and the fishery (Gibson et al. 1983, Strasdine 1988). Because SL limits have remained small over decades, the fishery targets predominately large juveniles. As a consequence, adults are relatively rare in the most heavily exploited habitats, seagrass and sand flats, occurring at depths of <10 m (Singh-Renton et al. 2006, Tewfik et al. 2019, Tewfik et al. 2021).

Here we summarize the population structure, based on shell length and lip thickness, as well as the decrease in shell length of mature, lipped conch at three major conch fishing areas (Turneffe Atoll, TU, TURF area 6; Lighthouse Atoll, LH, TURF Area 7 and Glover's Atoll, GR, TURF Area 8) (Fig. 14) (Tewfik et al. 2019, Tewfik et al. 2021). The three atolls of Belize are the most productive fishing grounds in the country, accounting for the majority of national conch and Caribbean spiny lobster landings for domestic and export markets. It is immediately apparent that the minimum shell length regulation (> 178 mm) allows many juveniles (no lip) and sub-adults (thin lips) to be harvested, while also allowing a number of small, but sexually mature, adults (lip thickness > 10 mm) to remain unharvested (Fig. 14a-c) due to their illegal shell length. The SL distribution at TU, LH and GR, respectively shows that 22-27% of juveniles are larger than 178 mm SL, and thus subject to legal harvest, and that 92%, 72% and 85% of sub-adults (thin lips) at TU, LH and GR are subject to legal harvest. Therefore, a large proportion of the population that is not yet mature and has never spawned is removed prematurely. This incurs a proportional loss in meat yield, which is lower in immature conch, and potentially a complete loss of fecundity (Stoner et al. 2012b). A shell lip regulation (e.g., > 10 mm) would greatly increase the catch of sexually mature conch, increasing both meat yield per individual caught and population fecundity (Fig. 14 d-f).

The rarity and low abundance of adults in fished areas (Tewfik et al. 2019, Tewfik et al. 2021) has led to the speculation that the conch population is supported by a deeper-water (> 25 m) spawning stock. Support of this comes from the general trend of conch to move, where access is possible, into deeper water as they mature (Doerr and Hill 2013) and the occurrence of conch at these depths in other locations in the region (Garcia-Sais et al. 2012, Boman et al. 2021). This assumption is equivalent to a management strategy of putting all your eggs in one basket without knowing where the basket is located and the number of eggs it contains. Nevertheless, this concept has only limited empirical evidence from Belize based on sampling at only 7 forereef sites to 18 m depth (Singh-Renton et al. 2006). In contrast to shallow areas these deep sites contained mature and larger adults, but densities, even in reserve areas, were not high relative to management guidelines (Appeldoorn 2025). Similar results have been reported for conch in the forereef of Glovers Atoll (Tewfik et al. 2019). These results only give limited support for the deep-water spawning stock hypothesis, further emphasizing the need for management measures targeting appropriate minimum sizes and the maintenance of higher adult densities overall to maintain spawning activity.

The existing minimum size regulation of 178 mm SL (and accompanying 85 g market clean meat limit) has been in place for more than 45 years, but available data clearly show this limit to be ineffective in maintaining population stability and sustaining productivity (Tewfik et al. 2019). In addition to the issues discussed above, the significant proportion of large juveniles and sub-adults, likely the fastest growing phenotypes, being harvested with the current size limit is leading to a truncation of the SL size distribution of flared shell-lip conch (i.e., subadults and adults) over time (Tewfik et al. 2019, Tewfik et al. 2021). The proportion of adults under 178 mm SL across atolls is 17%, 74% and 18% for TU, LH and GR, respectively. This shell length truncation is clearest at Glover's Atoll based on a 15-year monitoring program. There, in the highest density and most fished conch habitats (sand flats, patch reefs), the shell length of lipped conch (sub-adults and adults) has declined by 13% and 15%, respectively, in the two habitats (Tewfik et al. 2019) (Fig. 14 g). For reference, a 15% decline in SL would represent 44% decline in tissue weight (meat + visceral mass, including the gonad) at the time of the onset of shell lip formation (Appeldoorn 1988). Similar patterns are seen at Turneffe and Lighthouse atolls with 11% and 7% declines, respectively, in shell length of lipped individuals over shorter monitoring periods (Fig. 14h, i). It should be noted that at Lighthouse Atoll "samba" conch, a thick-lipped, small SL phenotype that are confirmed to have smaller tissue and gonads and therefore lower meat yields and fecundity (Stoner et al. 2012b, Kough et al. 2017), constitute a staggering 74% of all adults. This may indicate that the SL truncation impacts of the minimum SL regulation may have been ongoing well before monitoring programs began. These trends suggest that management needs to change. Refinement of the individual size-based regulations for conch in Belize to a lip thickness > 10 mm (with a corresponding 192g market clean meat) would increase individual meat yield and fecundity (Tewfik et al. 2019) and protect densities in spawning areas. Spawner density should be > 50 adults/ha to help avoid apparent allée effects for this species (Stoner et al. 2012a, Tewfik et al. 2019).

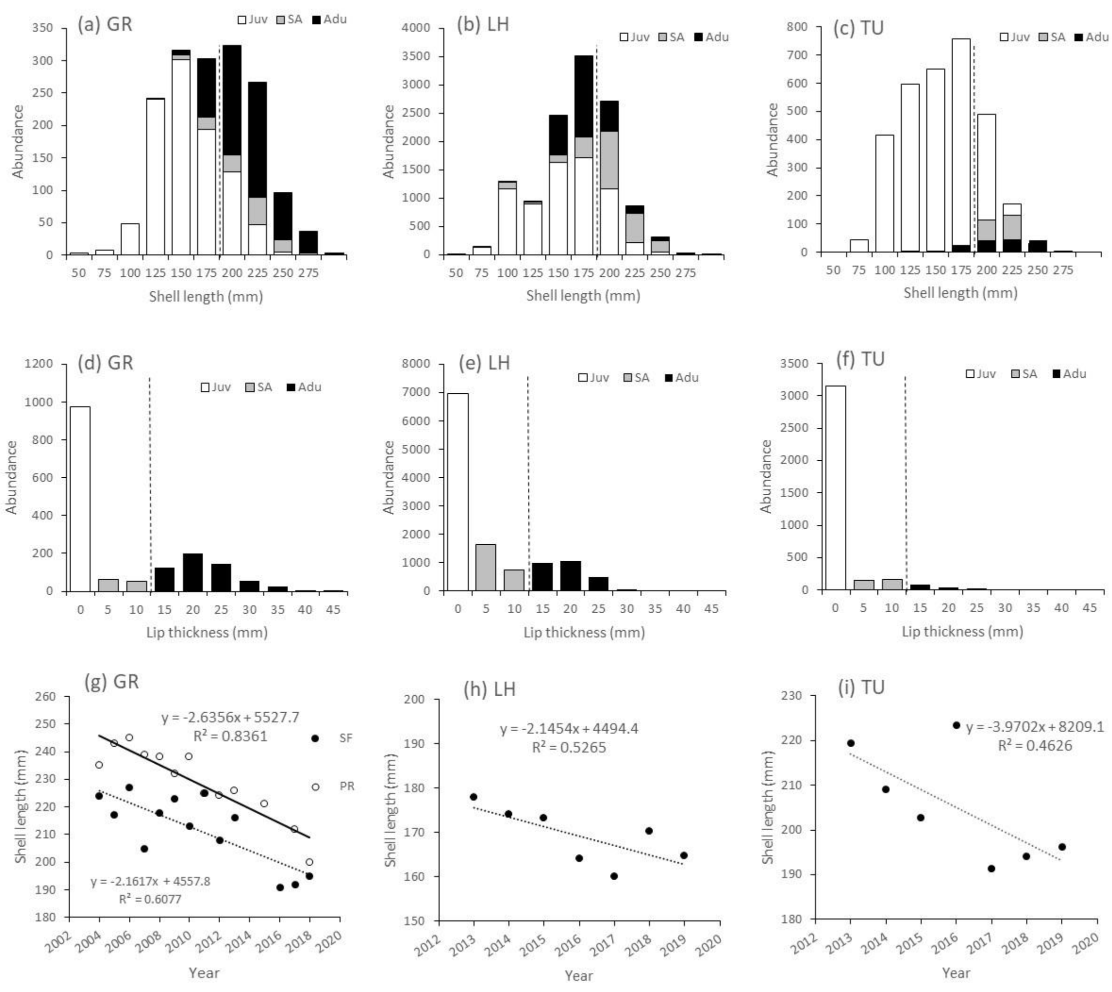


Figure 13: Belize queen conch (*Aliger gigas*) population structure (shell length, a – c; and lip thickness, d - f) and declines in sub-adult/adult (lipped) shell length over time (g – i). Note that dashed lines indicate legal minimum size (178 mm shell length, a - c) and lip thickness of maturity (10 mm, d - f). Glover's reef (GR): (a,d) population structure 2015 – 2018 across 4 habitats, sandflats (n = 1169); patch reefs (N = 330); seagrass (i.e. lagoon floor, n = 78); and fore-reef (n = 71), total n = 1648. (g) Shell length decline (lip 1 mm or greater) in replenishment zone only 2004 – 2018, patch reef (n = 1215), sand flats (n = 1458). Lighthouse atoll (LH): (b, e) population structure 2013 – 2019 in across habitats, sandflats (n = 3618), seagrass (n = 8622), total n = 12240. (h) Shell length decline (lip 1 mm or greater) in replenishment zone only, 2013 – 2018 sand flats (n = 1724). Turneffe atoll (TU): (c, f) population structure 2013-14, 2016-17 across 4 habitats, sandflats n = 843, patch reef n = 348, seagrass n = 2377, fore reef n = 41, total n = 3609. (i) Shell length decline (lip 1 mm or greater) in general use zone only, Sand Flats (n = 146) (2013 – 2019). All data presented in Tewfik et al. 2021.

4.2 Caribbean spiny lobster (*Panulirus argus*)

Spiny lobster catches provide critical income to fisheries-dependent households throughout the Caribbean where lobsters are harvested using traps, artificial shelters (i.e., casitas, shades) as well as snaring, hooking and spearing by breath-hold divers (Cochrane and Chakalall 2001, Tewfik and Bene 2004, Babcock et al. 2015). The fishery for spiny lobster is the most valuable wild capture fishery in Belize with export earnings estimated at 10 million USD annually as mostly exported tails, whole animals and head meat (Greers et al. 2020). However, it has been reported that average sizes of lobsters appear to be declining and that some individual fishers have insufficient catch to support their domestic needs (Greers et al. 2020). This apparent overfishing may be the result of possible overcapacity in numbers of fishers and gear (e.g., trap and artificial shelter density). Observations at Turneffe Atoll, before the designation of the reserve (2012), indicated a 70 percent decline in lobster sales to cooperatives between 2004 and 2009. The long-held minimum size limit of 76 mm carapace length may have been inadequate to protect immature lobsters (Greers et al. 2020).

A local study undertaken more than two decades ago noted that there was a significant discrepancy between the legal tail mass minimum of 113.4 g (4 oz), on average corresponding to a CL of 76 mm (3 inches), and CL minimum of 82.5 mm (3.25 inches) at the time (Wade et al. 1999). The result was that lobsters that did not meet the 82.5 mm CL regulation for whole lobster export could be processed to tails only, as they generally are for a large portion of current exported product, and thus satisfy the 113.4 g tail minimum (Wade 1999). Less than 15% of lobster were mature at 76 mm CL. The recommendation at the time was to retain the 82.5 mm CL minimum and increase the tail minimum to 154 g (5.4 oz) to avoid catching lobsters before they have a chance to reproduce (Wade et al. 1999). Despite a number of concerns the 76 mm CL minimum has been used following the recommendations of a regional working group on spiny lobster management to "protect the spawning stock" (FAO, 2007; Gongora, 2010). Most recently, a study using changes in reproductive-related structures (e.g., pleopod setae length, gonopore diameter) over a wide range of harvested sizes of lobster examined on fisher boats and in a large fishing cooperative concluded that CL at 50% maturity differed between males (98 mm) and females (86 mm), and was higher for both sexes than the existing legal minimum of 76 mm (Tewfik et al. 2020). The study recommended an increase in the legal minimum to 86 mm CL. This would more effectively protect many immature lobsters, future fecundity of the spawning stock and recruitment to this very lucrative fishery. Amended regulations have now increased the CL to 82.5 mm (3.25 inches) and tail mass of 127.6 g (4.5 ounces), which is better, but it means that the tail mass minimum again does not match CL minimum and will allow many immature lobster to enter the market legally as tails (Wade et al. 1999, Tewfik et al. 2020, Anonymous 2021).

Tewfik at al. (2020) also examined a number of fisheries indicators for lobster populations across three fishing grounds (TURF Area 2 – North central/Belize City; Area 3 – South central/Dangriga; Area 8 - Glover's atoll Fig. 3). These three TURFs represent the effort of more than 2000 fishers or 2/3 of the entire licenced fishing community and more than 4200 km^2 of open fishing area. The proportion of mature lobster in the catch (Pmat) for both sexes increased from TURF 2 to 3 to 8 but were all less than 55% and as low as 15% for males in area 2 (Fig. 15a). These trends were mirrored in two other indicators. The ratio of fishing mortality to natural mortality (F/M) was well above 1.0 for all three grounds, indicating that overfishing was occurring, with the lowest values found at Glover's (Fig. 15b). Similarly, the spawning potential ratio (SPR) was well below 20% for all grounds, indicating that the spawning population was overfished, with Glover's Atoll again being in the best condition relative to the other two TURFs examined (Fig. 15c). These trends can be related to the proportion of no-take area (i.e. replenishment zone) to area of the entire fishing ground (i.e., TURF) as well as the time since establishment of the no-take areas (Fig. 15d). This may provide a reasonable gauge of the potential effectiveness of the protected area (e.g., spillover to fished areas) on the resident lobster populations. Glover's Atoll has the largest and oldest (30 years) relative no-take area and the best values for Pmat, F/M, and SPR despite overfishing being a problem across all three fishing grounds examined (Tewfik et al. 2020). The fishing area (TURF 2) closest to Belize City showed the opposite trend, indicating that higher fishing pressure may be correlated to proximity to urban centers for Belize. It should be noted that the density of licensed fishers over the open fishing area of each TURF is highest for Glover's Atoll (area 8, 0.73 fishers/km^2) with decreasing densities in area 3 (0.57 fishers/km^2) and area 2 (0.37 fishers/km^2) (Tewfik et al. 2020). This may indicate that the proportion of good lobster habitat is highest at Glover's, supporting higher fisher density, but also that well-managed sites and community engagement over time can yield positive results even when fishing pressure is high. Finally, a time series of mean lobster tail masses over a decade in Area 3 and Area 8 indicate declines (Fig. 15e), and thus decreased overall catch, as corroborated by the testimony of local lobster fishers (Greers et al. 2020).

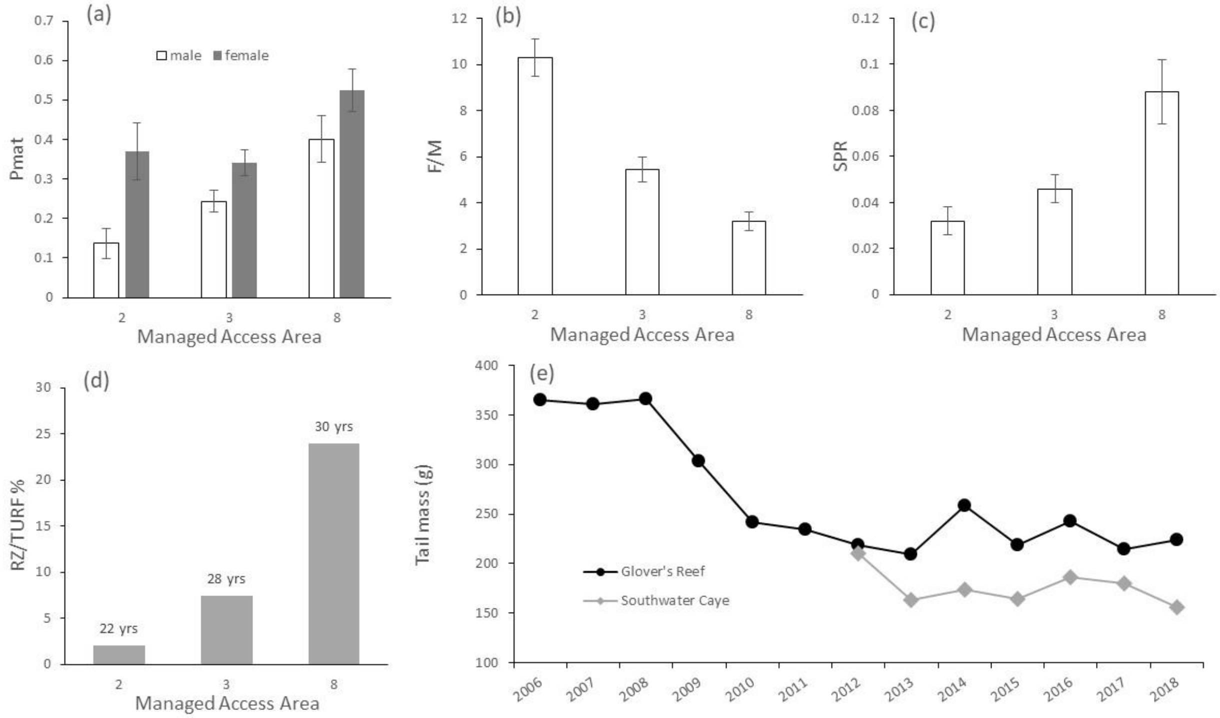


Figure 14: Belize Caribbean spiny lobster (*Panulirus argus*) fisheries indicators (a, proportion mature, Pmat; (b) ratio of fishing to natural mortality, F/M; (c) spawning potential ratio, SPR), (d) ratio of replenishment zone (RZ) (i.e. no take) to TURF area, RZ establishment time and (e) tail mass (g) over time at 3 managed access (TURF) areas (2, 3, 8, see Fig. 3).

4.3 Groupers (Serranidae)

The extraction of large groupers, often taking place at well-known fish spawning aggregations (FSA) using hook and line (88%, Fig. 12), are amongst the first large-scale modern commercial fisheries in Belize dating back to the earliest part of the 20th century (Craig 1966, 1969). The best known and most caught species was the Nassau grouper (Burns-Perez and Tewfik 2016, Phillips 2024), now considered critically endangered (Sadovy et al. 2018). After a century of continuous commercial harvest, extensive surveys revealed that only two of nine traditional Nassau grouper FSAs numbered in the thousands (NE Point at Glover's Atoll and Sand-bore Caye at Lighthouse Reef Atoll) (Fig. 16), with the remaining sites having very low numbers or no fish at all (Paz and Grimshaw 2001, Sala et al. 2001, Burns-Perez and Tewfik 2016) (Fig. 16). Despite directed management measures (see below), there is little evidence of FSA recovery, and the pattern of decline continues today. At Glover's Atoll an 85% decline has been observed at the only known FSA as well as declines observed in general reef surveys and in the catch of licenced fishers over the last 20 years (Phillips et al. 2025). Similarly, other large serranids including goliath, black, and yellowmouth (*Mycteroperca interstitialis*) groupers were also targeted over an expanding spatial range, seasons, and gear types (e.g., spears), despite seasonal closures and fully protected FSA sites (Heyman and Kjerfve 2008, Paz and Sedberry 2008). At the Gladden Spit FSA, Nassau and other large groupers (e.g., black, yellowfin - *Mycteroperca venenosa*) aggregate with overlapping timing, and all show patterns of decline beginning in the mid-2000s after some earlier recovery (Heyman and Wade 2007, Heyman and Kjerfve 2008, Tewfik 2023). However, based on the most recent data, the aggregations of black, Nassau, and yellowfin grouper (endangered, IUCN) had effectively disappeared at the Gladden Spit FSA by 2017 (Tewfik 2023).

The focus of fishing activity at FSAs has serious implications for future reproduction and recruitment of grouper populations (Sala et al. 2001, Sadovy and Domeier 2005, Sadovy de Mitcheson et al. 2013, Hixon et al. 2014). Specific management and conservation measures targeting Nassau grouper and implemented by the early 21st century have included minimum and maximum (to protect mega-spawners) harvest sizes, a closed season, and full-time closures of many FSA sites. The latter two have also afforded protection to other species (Heyman and Wade 2007, Burns-Perez and Tewfik 2016). However, recent stock assessments (Fig. 7), overall catch data collections across the BBR (Fig. 8, 13, 17b), and fishery-independent visual surveys outside FSAs across a range of grouper life histories indicate that overall grouper abundance is low (Fig. 11, 17c) (Tewfik et al. 2022, Palomares et al. 2023, McField et al. 2024b). This is especially true for large species (e.g., goliath, black, Nassau), which have been found to be disproportionately immature (< Lm) when caught (Fig. 13, 17b). A size-spectra shift in extractions towards the remaining smaller species (red hind, *Epinephelus guttatus*; coney, *Cephalopholis fulvus*; graysby, *C. cruentatus*) appears to be taking place (Fig. 13, 17b) (Tewfik 2018, Tewfik et al. 2018, Tewfik et al. 2022, McField et al. 2024b). The biomass of these smaller predatory serranids increased dramatically (880%) over a seven-year period (Mumby et al. 2012). This change has been attributed to a release from predation and constraints to foraging behaviour by large serranids that are now overfished and rare, as well as reduced populations of large snappers and sharks. The Nassau grouper (Lmax 122 cm) (Sadovy et al. 2018) has now been effectively replaced as the most caught grouper in Belize (1960s) by the much smaller red hind (Lmax = 76 cm) (Fig. 17) (Tewfik et al. 2022). The loss of large groupers, a mainstay of Belize fisheries for decades, will have a fundamental impact on the resilience of reef complex communities, livelihoods and food security in Belize.

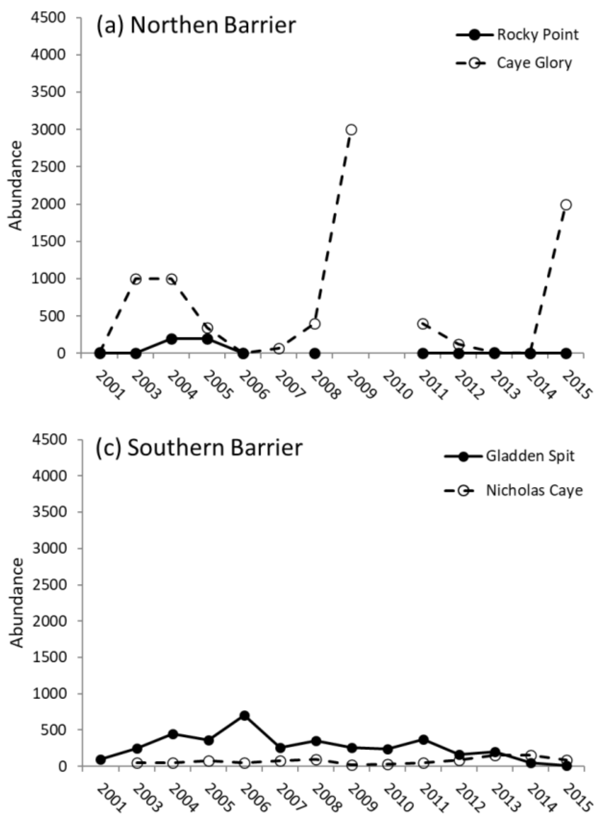


Figure 15: Monitoring data (2001 – 2015, excluding 2002) of eight fully protected Nassau grouper (*Epinephelus striatus*) spawning aggregation sites across Belize divided into geographic sub-areas: (a) Northern barrier reef; (b) Turneffe Atoll (c) Southern barrier reef and (d) Outer Atolls (Lighthouse and Glover's). Data source: Belize Spawning Aggregation Database. Redrawn from Burns-Perez and Tewfik 2016.

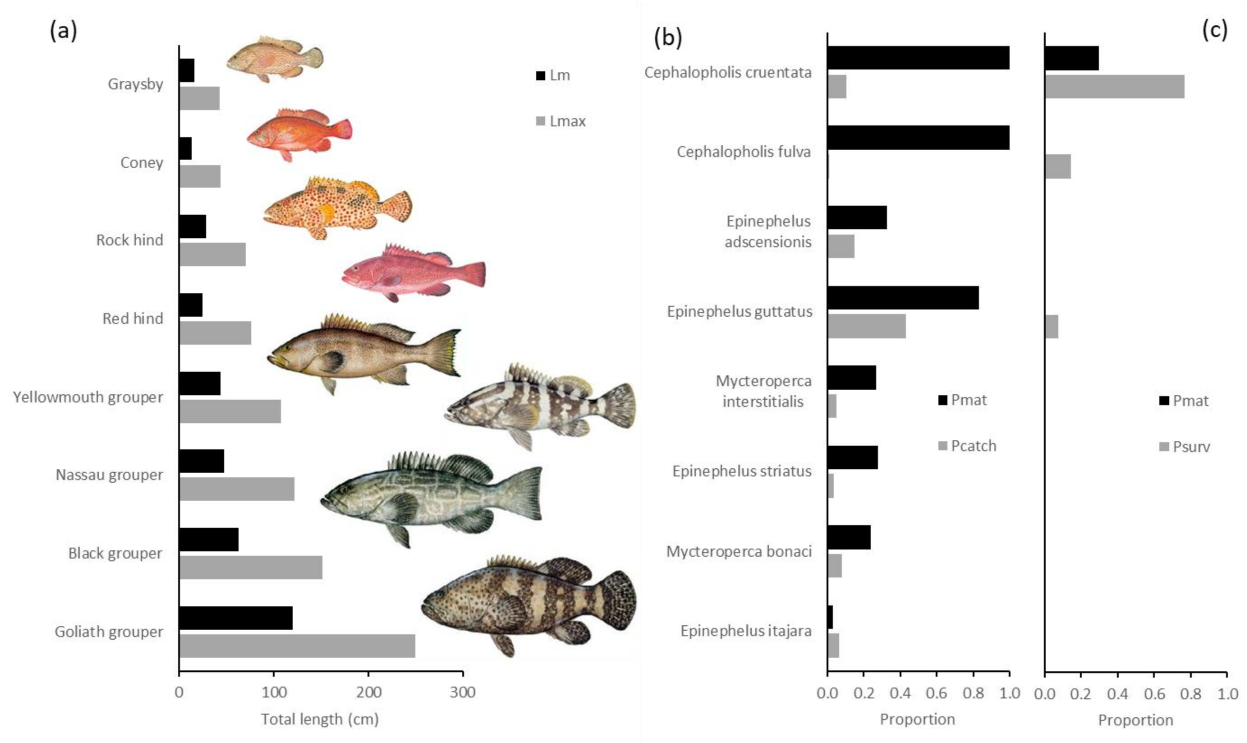


Figure 16: Life history, catch and fishery-independent survey data for groupers in Belize. (a) Size at maturity (Lm) and maximum size (Lmax) for species (Tewfik et al. 2022). (b) proportion of the catch (measured indiv.) that are mature (Pmat) and proportion of observed species from all groupers observed (n = 516) (Pcatch) (Tewfik 2018 a,b; Tewfik et al. 2022). (c) In-water survey Pmat (based on count of mean bin size) and total count data (Psurv) (McField et al. 2024b).

4.4 Snappers (Lutjanidae)

Large snappers have always been amongst the favoured species of Belizean fishers and consumers, along with large groupers, and are the dominant family caught across most of the country (45.6%) (Table 1, Fig. 8, 10) (Heyman and Graham 2000, Phillips 2024). The catch of snappers ranged between 31 – 59% of catch across TURFs, mostly using hook and line (69%) (Fig. 12) and amounting to more than 17 times the number of groupers caught overall (Tewfik 2018, Tewfik et al. 2018, Tewfik et al. 2022). However, snappers, comprising 14 documented species (Tewfik et al. 2022), are experiencing similar patterns of decline to those observed in groupers. This was already noted in the mid-20th century with the loss of a mutton snapper FSA at Long Caye, Belize, after fishers targeted fish along their reproductive migratory pathway to the FSA (Craig 1966). More recent evidence suggests historical declines in both catch and mean length of mutton snapper at the Gladden Spit, the largest known legal mutton snapper FSA fishery on the Mesoamerican Barrier Reef (Graham et al. 2008). In addition, significant illegal activities have led to the under-reporting of catches for cubera snapper (Heyman et al. 2005). More than two decades of FSA surveys at Gladden Spit observed declining numbers of the largest species (cubera, dog (*Lutjanus jocu*) and mutton) in both maximum counts and sizes (Tewfik 2023).

Harvesting regularly at FSAs is taking its toll on the broader populations of snappers across Belize. Recent data across a range of snapper life histories (Fig. 18) suggests that snapper populations over that last 15 years are generally in fair to poor condition (Fig. 11), large (cubera, dog) and medium sized (mutton, gray (*Lutjanus griseus*), yellowtail) species are often immature (< Lm) in the catch (Fig. 13, 18b) and rare and immature in visual surveys outside of FSAs (Fig. 18c) (Tewfik et al. 2022, McField et al. 2024b).

Smaller snapper species have become an increasing proportion of the remaining catch, and are also mostly mature (Fig. 13, 18b). In a clear sign of size-spectra shift, the lane snapper (Lmax = 61 cm) has become the most caught (15.5%) fish in Belize, now landed in a significantly higher proportion than the formerly dominant and larger mutton snapper (Lmax = 94 cm) at about 5% of the total catch (Tewfik et al. 2022). The hogfish, *Lachnolaimus maximus* (Family Labridae), while not a true snapper but often referred to as one, is a prized species, including recreationally. It accounts for 2.2% of the total catch, but only 26% of landed individuals are mature (Tewfik et al. 2022). As with groupers, the loss of large snappers will have significant impacts on reef complex resilience and livelihoods in light of other disturbances such as nutrient pollution and climate change.

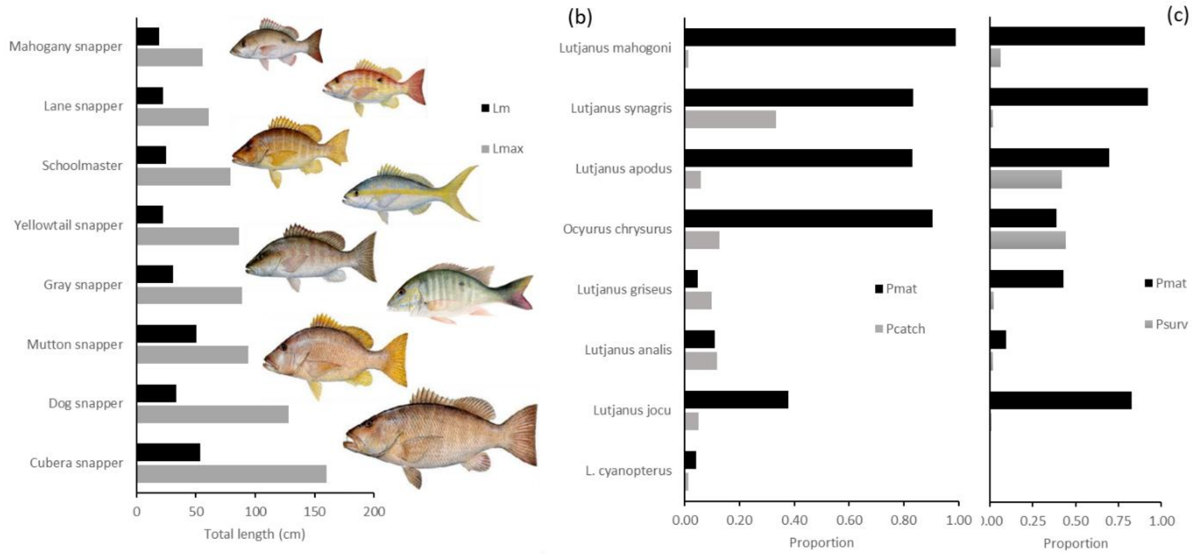


Fig. 17: Life history, catch and fishery-independent survey data for snappers in Belize. (a) Size at maturity (Lm) and maximum size (Lmax) (Tewfik et al. 2022). (b) Proportion of the catch (measured indiv.) that are mature (Pmat) and proportion of observed species from all snappers observed (n = 9045) (Pcatch) (Tewfik 2018 a,b; Tewfik et al. 2022). (c) In-water survey Pmat (based on count of mean bin size) and total count data (Psurv) (McField et al. 2024b).

4.5 Other Boney Fish (Carangidae, Haemulidae, Sparidae, Scombridae)

A number of other fish families provide significant contributions to modern bony fish catches in Belize. The jacks (Carangidae) are reef associated, pelagic piscivores that may school regularly in large numbers and account for almost 13% of the total catch observed between 2016 and 2020 (Fig. 8), being mostly caught using hook and line (51%) and gillnets (30%) (Fig. 12d) (Tewfik 2018, Tewfik et al. 2018, Tewfik et al. 2022). Of the 9 species documented, some of the most frequently caught are larger species (e.g., rainbow runner (*Elagatis bipinnulata*), crevalle jack (*Caranx hippos*)) that have low proportions of mature fish (> Lm), with smaller species (e.g., bar jack, *Caranx ruber*) showing much higher rates of maturity (Fig. 13, 19a). It appears that the availability of large jacks (e.g., crevalle, horse-eye) in proximity to shallow water reefs makes them important components of the commercial catch, possibly as a replacement for now rare large grouper and snappers.

The mostly smaller, invertivorous reef-inhabiting grunts (Haemulidae) account for 8.2% of recent catches (Fig. 8) (Tewfik 2018, Tewfik et al. 2018, Tewfik et al. 2022) but do not rank highly in value for commercial purposes. However, grunts are popular for domestic consumption (Greers et al. 2020) and may be critical for subsistence with their high abundance and ease of catch using inexpensive hook and line (74% of landings, Fig. 12e) on shallow and proximate patch reef habitats. Amongst the grunts, 11 species were observed, with the largest Caribbean species, the white margate (*Haemulon album*) (Lmax 79 cm), having only 38% of individuals being landed mature (> Lm) and representing a very small proportion of the overall catch (Fig. 19b). A small species, French grunt, are 97% mature when landed but constitute a low proportion of the catch, while medium sized grunts (white (*Haemulon plumierii*) and blue-striped (*Haemulon sciurus*)) display intermediate proportions landed mature and accounting for most of the catch of grunts (Fig. 19b).

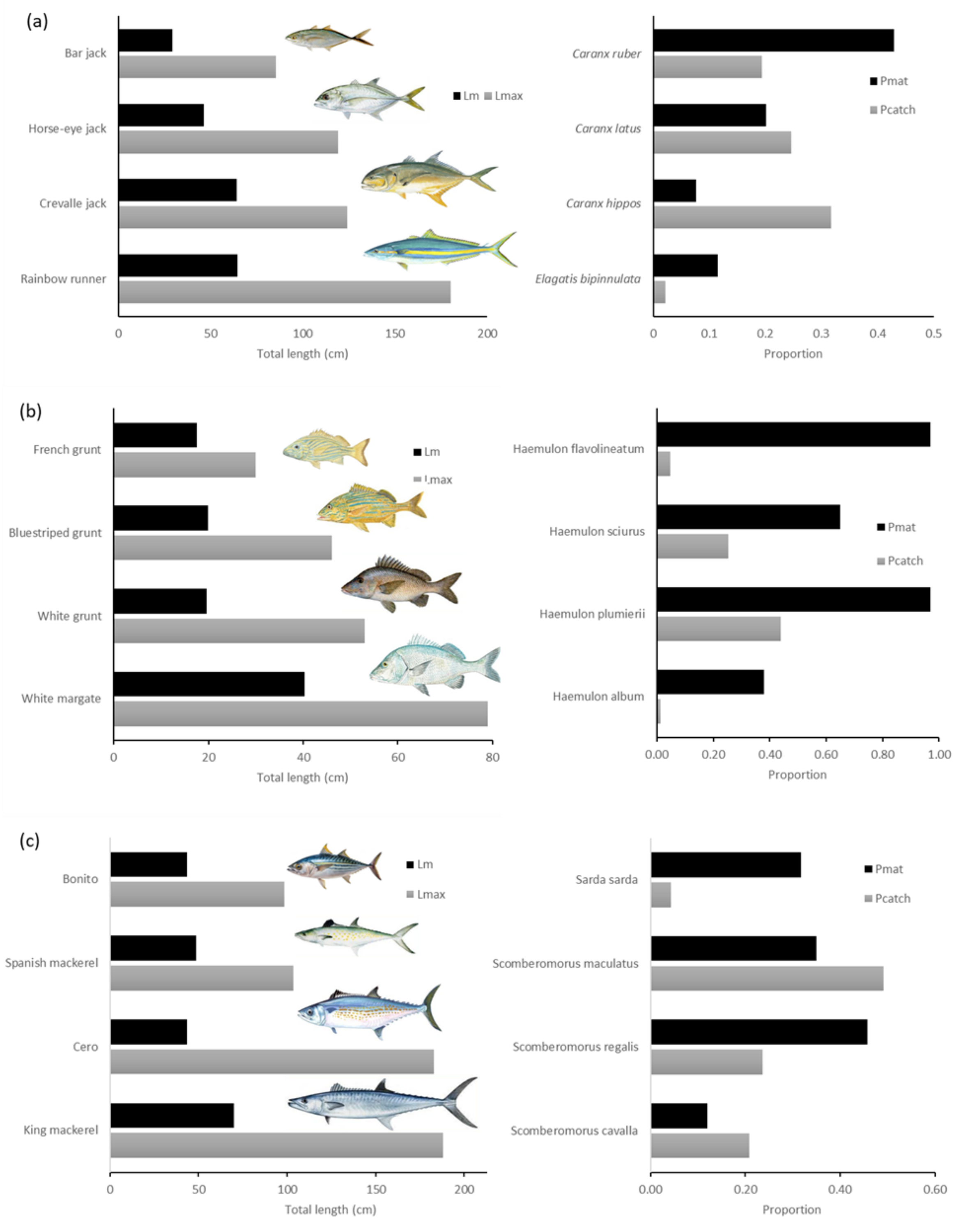


Fig. 18: Life history, catch and fishery-independent survey data for (a) jacks (Carangidae) (n = 2501), (b) grunts (Haemulidae) n = 1635), and (c) mackerels (Scombridae) (n = 1017) in Belize. Size at maturity (Lm) and maximum size (Lmax) (Tewfik et al. 2022), proportion of the catch (measured indiv.) that are mature (Pmat) and proportion of listed species from all fish observed in the family (Pcatch) (Tewfik 2018 a,b; Tewfik et al. 2022). (c) In-water survey of individual bin-sized giving Pmat and count data (Psurv) (HRI 2024).

The invertivorous and reef-associated (benthic) porgies (Sparidae) occupy a similar ecological niche as grunts and show similar patterns in the landings. They constitute five documented species (e.g., jolthead (*Calamus bajonado*), pluma (*Calamus pennatula*), sheephead (*Calamus penna*)) (Tewfik et al. 2022), and 2.9% of the catch (Fig. 8), being caught mostly with hook and line (Fig. 12g). The proportion mature in the catch ranges from 38 to 74%, with larger species showing lower proportions. Again, despite their lower commercial appeal, they may be important for subsistence or as cheaper alternatives to small snappers and groupers available in local markets (Greers et al. 2020).

Finally, the pelagic-neritic Scombridae constitute 5.1% of the total catch (Fig. 8), using mostly hook and line (71%) (Fig. 12f), with the catch consisting mostly of mackerels (4 species, n = 1011) and a few tunas (frigate (*Auxis thazard*), yellowfin (*Thunnus albacares*), n = 8) (Tewfik 2018, Tewfik et al. 2018, Tewfik et al. 2022). The largest mackerel species (king) has the lowest proportion of mature individuals landed (12%), with medium sized species (cero, Spanish) being the most caught but with maturity of the catch being only 46% and 35%, respectively (Fig. 19c). The beginnings of a size spectra shift seem apparent within this family. These species are not generally associated with the shallow reef complex and may be targeted by a subset of skilled artisanal fishers that access these species in open-water (TURF area 9 – general to all commercial licences). They may also be caught by the broader fisher community as occasional transients over shallower habitats, edges of the fore-reef or in deeper areas of lagoons in TURFs under their specific licences. However, the large size of even the smaller species (e.g., bonito, Lmax = 98 cm) makes them valuable catch for the commercial markets, tourism and export.

4.6 Elasmobranchs

A Belizean shark fishery has existed for decades, with local fishing cooperatives consistently purchasing these products in the past (1960s to 1990s) and exports registered by the Department of Fisheries (Gibson et al. 2004, Graham 2007). Sharks are not commonly consumed in Belize so they are poorly represented in local market surveys (Tewfik et al. 2022). The blacktip (*Carcharhinus limbatus*) was the only shark species documented in urban fish market surveys across Belize with all individuals observed (n = 18) being immature (Tewfik et al. 2022). These landings may have been bycatch in gillnets before the ban. However, sharks are important as living resources to local SCUBA diving and snorkelling-associated tourism, supported by bans on harvesting of nurse and whale sharks (Graham 2008, Tewfik et al. 2022). Nevertheless, shark fishing does employ at least 75 specialized fishers (approximately 2.5% of all licenced fishers), working in groups of 3–10 within their permitted managed-access TURF areas, with landings and processing now occurring at isolated fishing camps on the mainland and cayes (Quinlan et al. 2021). Shark parts of primary interest are salted (meat) or dried (fins) and exported under licence, largely directed to Guatemala where many Belizean shark fishers also have family and business ties (Graham 2007, Zeller et al. 2011, Quinlan et al. 2021). In addition, liver oil and cartilage are traded locally for medicinal properties and jaws and teeth are sold as curios (Greers et al. 2020).

Given the importance of the Guatemalan trade route, it is significant that no legal exports from the shark fishery occurred during 2020 and 2021 due to the pandemic-related border closures (Tewfik et al. 2022).

Catch data collected at local shark fishing camps indicate that the Caribbean reef shark (*Carcharhinus perezi*) is the most caught species (32%) followed by sharpnose (*Rhizoprionodon* spp.), blacktip (*Carcharhinus limbatus*) and bonnethead (*Sphyrna* cf. *tiburo*) (Fig. 20) (Graham 2007, Quinlan et al. 2021). An additional 12 species have been observed in the landings, including a number of large species (e.g., tiger, *Galeocerdo cuvier*, great hammerhead, *Sphyrna mokarran*) listed on Appendix II of the Convention in the International Trade of Endangered Species of Flora and Fauna (CITES) (Fig. 20) (Jabado 2024). Recent trends in Belize shark landings (Quinlan et al. 2021) reflect an earlier assessment (Graham 2007). In the most recent landings study, using fisher submitted anal fins to estimate total length, the top four landed species all include immature individuals (Quinlan et al. 2021). The larger Caribbean reef and blacktip were mainly caught immature ($P_{mat} \leq 0.1$), while the majority of the smaller sharpnose and bonnethead landed were above the size of maturity ($P_{mat} = 0.7$ and 0.6) (Fig. 21). The calculated overfishing indices (L_x/L_m) for the most caught species based on published frequency distributions and length at maturity studies presented in Quninlan et al. (2021) (Fig. 21) were 0.53 (reef, $L_{max} = 300$ cm), 0.68 (blacktip, $L_{max} = 286$cm), 0.95 (bonnethead, $L_{max} = 150$ cm) and 0.98 (sharpnose, L_{max} 113 cm) indicating, as with bony fish (Tewfik et al. 2022), that overfishing generally increases with increasing species size and trophic position (Fig. 6). The relative productivity (intrinsic rate of population increase) and associated resilience to fishing pressure of the main exploited shark species (i.e. smaller – sharpnose, bonnethead) exploited in Belize is thought to be high even as such small species, bonnetheads, have been nearly extirpated due to fishing in Brazil (Greers et al. 2020). However, conclusion on relative resilience (based on Fig. 14, Greers et al. 2020) did not include an assessment for the most commonly caught (32%) and relatively large carcharhinid, the Caribbean reef shark. Caribbean reef sharks reproduce with a biennial reproductive cycle where gestation occurs for approximately one year, producing three to six shark pups which is indicative of low reproductive productivity (Carlson 2021). In addition, where it is not protected, Caribbean reef shark population reductions of 99% over three generation lengths (29 years) have been documented (Carlson 2021).

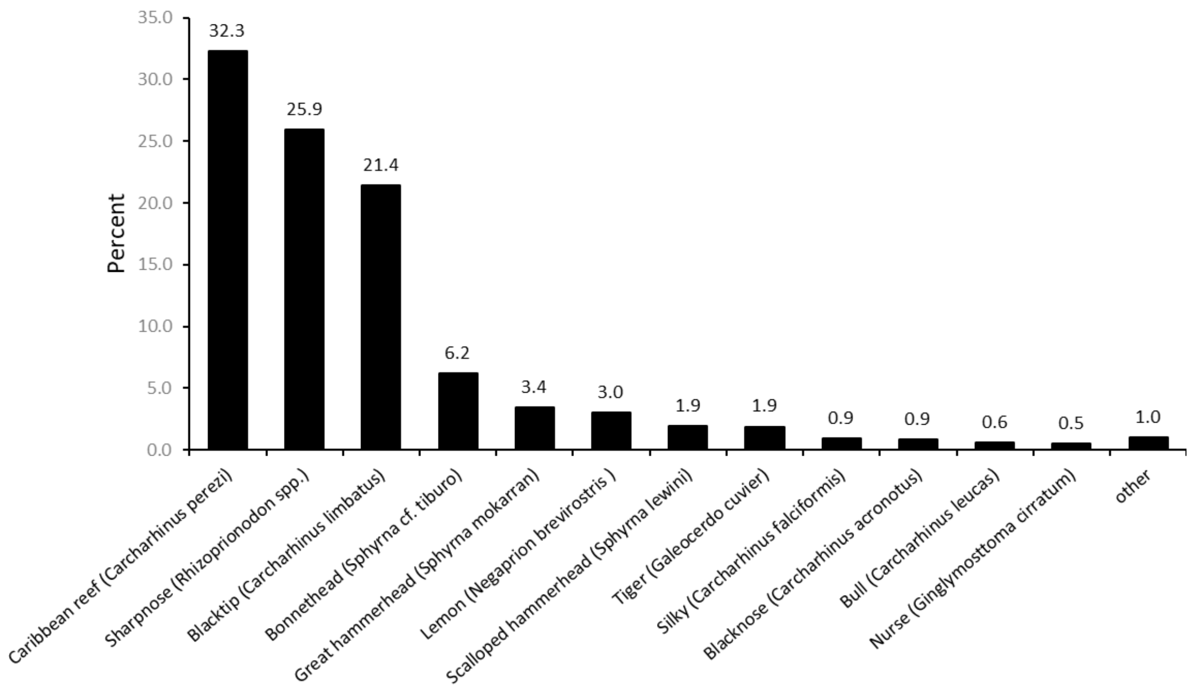


Figure 19: Percent commercial landings (n = 1969) of sharks in Belize from combined from two studies (Graham 2008, Quinlan et al. 2021). Other category (n = 20) includes Atlantic sixgill (*Hexanchus vitulus*), dogfish (*Mustelus spp.*), night (*Carcharhinus signatus*), shortfin mako (*Isurus oxyrinchus*) and spinner (*Carcharhinus brevipinna*) sharks. Note that landings of nurse sharks ceased once designated as fully protected with whale sharks in 2011.

The Caribbean reef shark, is also the best studied shark species in Belizean waters, including through the use of experimental long-lines, acoustic tracking and baited remote underwater video (BRUVs) (Pikitch et al. 2005, Bond et al. 2012, Baremore et al. 2021) (Fig. 22). This species may provide important insights into the current state of the wider suite of harvested sharks. Size frequency data from Glover's and Lighthouse Atolls indicates a similar distribution of total length, from 60 to 260 cm, with many individuals appearing below the Lm of 182 cm (Fig. 22) These juveniles appear disproportionately in landings data and may indicate a directed or inadvertent targeting of smaller size classes due to chosen gear, bait, season or fishing area. It is also noteworthy that stable isotope analyses determined the trophic position of Caribbean reef sharks to be similar to other large-bodied teleost piscivores (barracuda, Nassau, black grouper) that were previously thought to be important prey for reef sharks (Bond et al. 2018). Reef sharks do not occupy an apex position in these food webs but are likely still critical couplers of food web compartments across large areas (e.g. between atolls) that tend to stabilize communities (Rooney et al. 2006). The competition with other large fished species (e.g., groupers and snappers) that seems apparent from the trophic information and co-occurrence over shallow and reef edge habitats could make them "bycatch" or incidental catch (e.g., blacktips seen at local markets) to the broader fishing community both licenced, but not for sharks, and poachers, especially at FSAs. Although an earlier study found that the reef shark population at Glover's Reef Marine Reserve appeared stable and evidence that marine reserves can be an effective

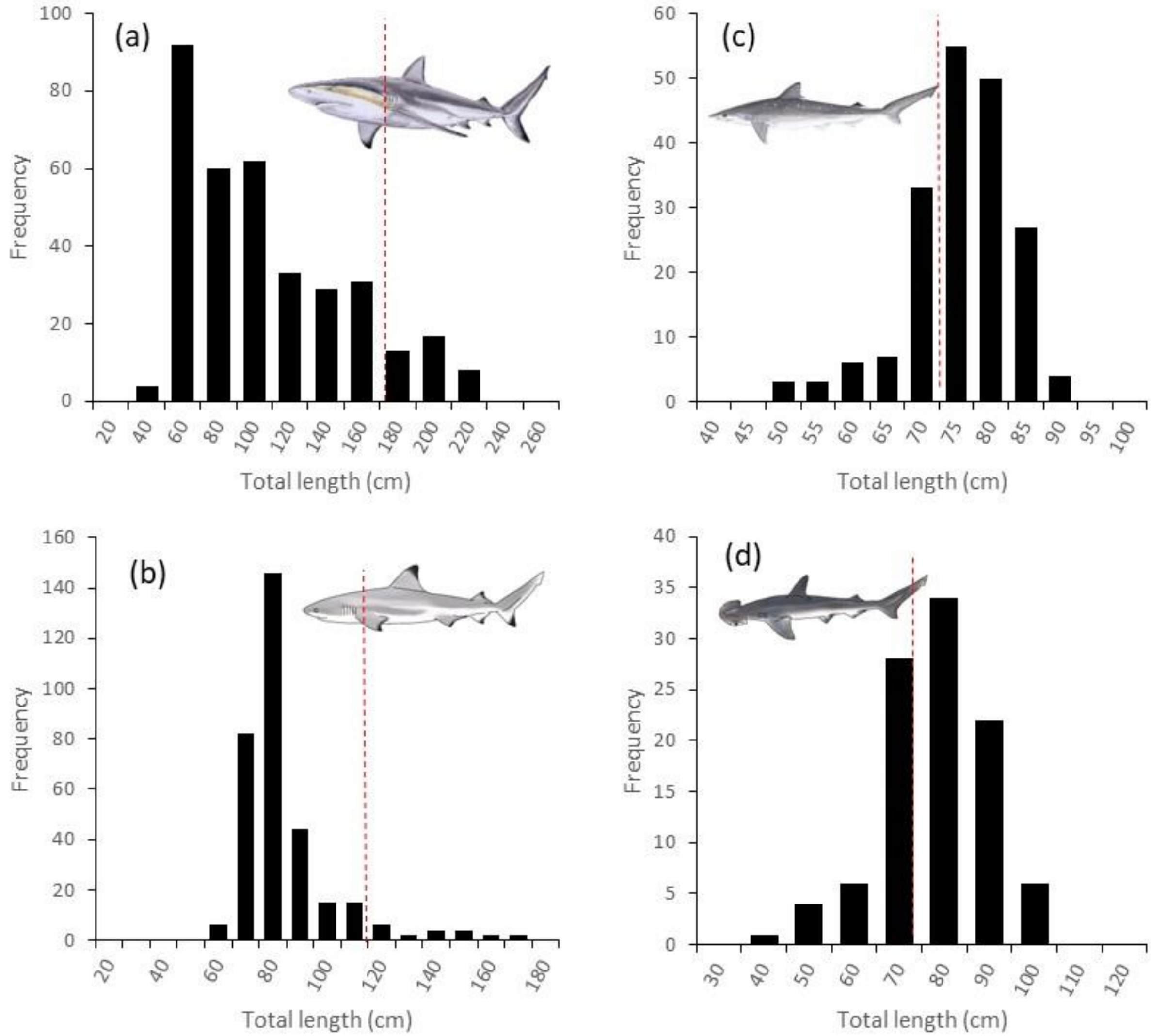


Figure 20: Frequency distributions of the mostly commonly caught shark species in Belize (redrawn from Qunilan et al. 2021). Length at maturity (red dashed line) was sourced from literature available in Quinlan et al. 2021: (a) Caribbean reef (*Carcharhinus perezi*), Tavares (2009); (b) blacktip (*Carcharhinus limbatus*), Carlson et al. (2006); (c) sharpnose (*Rhizoprionodon spp.*), Mattos et al. (2001), Carlson and Baremore (2003), and Motta et al. (2007); (d) bonnethead (*Sphyrna cf. tiburo*), Lombardi-Carlson et al. (2003).

conservation tool (Bond et al. 2017), more recent work indicates a decline in reef shark populations as a result of fishing along the edge of the Glover's atoll reserve, with *C. perezi* completely absent on BRUV stations (n = 38) in 2018 (Flowers et al. 2022). Based in part on these latest findings, shark fishing was banned within 3.2 km of all three atoll marine reserves (Anonymous 2021).

Rays have not been documented in any landings in Belize, and have been recently declared as fully protected (Anonymous 2020). However, it should be noted that as shark populations become depleted it appears that the southern stingray (*Hypanus americanus*) is seen more frequently on BRUVs (Bond et al. 2019). This may suggest a trophic cascade where populations of rays like *H. americanus* increase in the absence of their large predators (Caribbean reef sharks, scalloped and great hammerheads), or a behavioral response where rays move into new habitat when large sharks become rare. This in turn, given the demand for rays as food and bait in neighboring Guatemala (Hacohen-Domene et al. 2020, Castillo and Morales 2021), may give way to future illegal fisheries for currently protected Belizean rays being exported to Guatemala. Targeted fisheries for southern stingrays (*H. americanus*) and spotted eagle rays (*Aetobatus narinari*) already exist elsewhere in the region (Cuevas-Zimbron et al. 2011, Tagliafico et al. 2013).

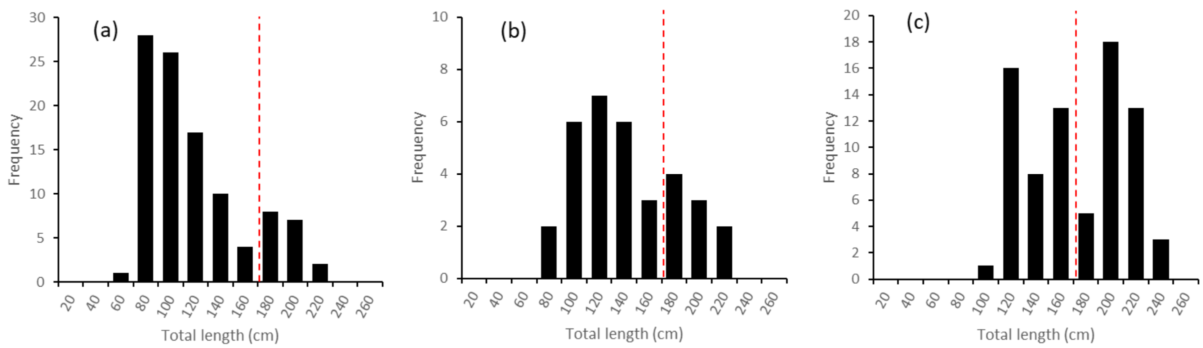


Figure 21: Frequency distributions drawn from available data for Caribbean reef shark (*Carcharhinus perezi*) in Belize collected during several studies at Glover's reef, (a) Pikitch et al. 2005 and (b) Bond et al. 2012 and (c) Lighthouse reef, Baremore et al. 2021. Dashed red line indicated the size of maturity (Tavares 2009 in Qunilan et al. 2021).

5. Legacy of Change

The Belize barrier reef complex and the human population of the area have had a long history of shallow-water resource use. The development of modern commercial fisheries and associated exports, beginning in the early 20th century, is fairly well documented and allows for an examination of why the resource base may now be eroded or overfished in the third decade of the 21st century. Signs of overfishing were already apparent by the early 1980s after decades of unmanaged extractions at FSAs exemplified by acute declines in Nassau grouper landings, and transitions to other large, aggregating groupers and snappers (Paz and Sedberry 2008). Other warning signs of the declining state of Belizean SSF include: (1) an assessment that some stocks were moderately exploited by the early 1990s (Koslow et al. 1994); (2) individual target (e.g., lobster, conch, lane snapper) sizes and numbers were in decline due to overfishing, smuggling and cross border fishing (Heyman and Graham 2000); (3) elasmobranchs that were once common showing dramatic declines (Graham 2007) including Caribbean reef sharks (Flowers et al. 2022); and (4) many commercial species were known to have been over-exploited including conch and lobster (Byron and Osipova 2013). A comparison of early and late 21st century (20-year period) bony fish catch (Heyman and Graham 2000, Tewfik et al. 2022) clearly indicates declines in most trophic groups and the top 5 families resulting in a greater than 14% decline in the Shannon diversity index (Fig. 10, Table 1). Though historically dominant, groupers (Serranidae) now represent less than 3% of the total catch even with the inclusion of the most heavily fished smaller species (e.g. red hind, *Epinephelus guttatus*) (Tewfik et al. 2022).

The eroded state of Belize's SSF may be described in more general terms as (1) lower proportions of mature individuals in landings, (2) a general size-spectra (relationship between organism size and abundance) shift within families due to lower availability and decreased landings of large species in favor of smaller species, and (3) a shift to smaller, lower trophic level, less valuable reef invertivores and medium-sized reef transients seen in local markets, the tourism trade and exports due to losses of traditional large and medium groupers and snappers. Declining maturity of the landings is supported by observations of smaller individuals being caught by several gears over time (Babcock et al. 2018) and that 51% of all individuals (N = 18383) measured onboard fishing boats, at local landing sites and in markets being juveniles across a broad swathe of species, most acute for larger species (Tewfik et al. 2022). Size-spectra shifts within families are well illustrated in landings for groupers and snappers, where high numbers of large species (e.g., Nassau grouper, Lmax = 122 cm; cubera snapper, Lmax = 160 cm) have been replaced, to a greater or lesser extent, by smaller species (e.g., red hind, Lmax = 71 cm; lane snapper, Lmax = 61) (Fig. 17, 18, 23) (Tewfik et al. 2022, McField et al. 2024b). General shifts in the abundance of smaller groupers (red hind, *Epinephelus guttatus*; coney, *Cephalopholis fulvus*; graysby, *C. cruentatus*) have been observed along with the loss of larger competitors as part of broader trophic shifts (Mumby et al. 2012), which include increasing landings of lower trophic level grunts and porgies as well as medium sized, transient mackerels and jacks (Greers et al. 2020, Tewfik et al. 2022) (Fig. 10, 19). Shark fishers have also noted shifts in species distribution and abundance, such as larger blacktips (*Carcharhinus limbatus*) that were once abundant being replaced by more common nurse sharks (*Ginglymostoma cirratum*), and Caribbean sharpnose (*Rhizoprionodon porosus*) (Graham 2007). These patterns of overfishing and associated trophic shifts indicate severely impacted populations, diminished opportunities for stock rebuilding and the overall erosion of critical trophic connections (Fig. 6) that are necessary to couple and stabilize food webs (Pauly et al. 1998b, Jackson et al. 2001, Valentine and Heck 2005, Rooney et al. 2006). This general pattern of overfishing indicates a need for targeted fishery management to reduce mortality on the depleted species, particularly on juveniles (e.g., size limits), because the existing system of license limitations (i.e. Managed Access), a complex marine protected area network, size limits and fishing seasons for only a few species, and gear restrictions have apparently not halted the decline and started stock rebuilding. As succinctly summarized by Allison et al. (1998), "Simply, the only successful control is where the scale of management is as large as the scale of the threat."

5.1 Marine Protected Areas Investment

From a fisheries management perspective the use of the closed area concept (Beverton and Holt 1957) potentially allows enhancement of fisheries in areas outside the no-take zone through the net export of larvae ('recruitment effect') and the net emigration of post-settlement animals ('spillover effect') (Rowley 1994, Roberts et al. 2001, Di Lorenzo et al. 2020). However, the effectiveness of protected area enhancement to fisheries is contingent on several factors (Di Lorenzo et al. 2020). First, the protected areas must be large enough to allow the build up of a critical level of mature and highly fecund individuals (i.e., spawning stock) to facilitate both the recruitment and spillover effects (Rowley 1994, Di Lorenzo et al. 2020). This will vary considerably depending on the life history and mobility of species as well as the level of fishing intensity. Second, the health of habitats must be high and sustainable to support both critical trophic interactions and fisheries production. The shallow, nearshore habitats and associated fishing grounds of the BBR complex are known to be impacted by climate change related threats and excessive nutrients due to various forms of land-based development (Lapointe et al. 2020, Lapointe et al. 2021, McField et al. 2024a). Third, the protected areas must be enforced to limit illegal fishing that compromises the build-up of mature biomass and, in the case of FSAs, allow unimpeded reproductive behaviour during limited periods at district and often remote locations (Sadovy and Domeier 2005, Heyman and Wade 2007, Phillips et al. 2025). The level of enforcement staff and assets needed to facilitate fisheries enhancement will vary depending on the size of the area, distance from shore and seasons of protection required (Sala et al. 2001, Gardner et al. 2003, Heyman and Wade 2007, Tewfik et al. 2017).

The spatial protections that are the primary focus of Belize's fishery management have had positive impacts (e.g. increased abundance, positive fisheries indicators) for some species, but they are not sufficient to make fisheries sustainable in the absence of more targeted fisheries regulations (Karnauskas et al. 2011, Bond et al. 2017, Tewfik et al. 2017, Tewfik et al. 2020, Flowers et al. 2022). The recent ban of any shark fishing throughout Glover's, Lighthouse and Turneffe Marine Reserves (Anonymous 2021), may be effective, along with the closed season for sharks and licence limitation, to improve the sustainability of the fishery. However, the general demise of significant large grouper and snapper populations even after decades of protection of FSA sites (Graham et al. 2008, Burns-Perez and Tewfik 2016, Tewfik 2023, Phillips et al. 2025) is perhaps the clearest evidence of the insufficiency of spatial protections alone. Recent ambitious expansion plans for marine protected areas under Blue Bond financing include 15% high protection zones that are considered as no-take and 15% medium protection zones where human activities are permitted and thought to be sustainable (Anonymous 2024). However, these plans (1) include large areas of deepwater offshore, requiring associated exploratory research efforts, are not traditional reef complex fisheries habitats; (2) may not adequately provide additional necessary, increased and ongoing enforcement and; (3) may not realize the promises of increased fish abundance (Thompson 2022), especially given past limitations. Also, considerations of the impacts of such MPA expansions to fisher livelihoods, gender and poverty must be explicitly considered in the design and management of these reserves (Barreto et al. 2020, Mizrahi et al. 2020). The majority of reviews of spillover assume abundance gradients imply benefits to the fishery with most providing no evidence (Di Lorenzo et al. 2020, Hilborn et al. 2025). To be an effective fisheries management tool MPAs must be imbedded in a suite of sustainable approaches such as science-based size and catch limits, engage fishers directly in management, and be highly enforced (Weigel et al. 2014, Di Franco et al. 2016, Prince and Hordyk 2018, Thompson 2022).

5.2 Underinvestment in Species-specific Harvest Management

Despite some of the successes ascribed to the large investment into Belize's marine protected areas network (i.e., reserves, replenishment zones, FSAs), the current level of fishery depletions indicates a need for more traditional and biologically based fisheries management strategies (e.g., species-specific minimum sizes, total harvest restrictions, closed seasons, gear restrictions). The recent "Managed Access" program aimed at controlling fishing effort using licence restrictions in TURFs (Karr et al. 2017) has been found to be primarily a re-packaging of traditional fishing areas with unenforced regulations and with the majority of stakeholders having a negative response on implementation (Wade et al. 2019). While having great promise, it appears that "Managed Access" has been overwhelmed by increasing numbers of participants and shifting of effort to remaining most productive fishing grounds at the three atolls (e.g., licences issued in Area 8: 168 in 2019, 532 in 2024) (Phillips et al. 2025).

The broader inclusion of species-specific management using life history traits would allow focus on commercially valuable species of conservation concern and broader biodiversity protection and contribute to a more holistic sustainability strategy. The low trophic level, but fisher income critical, populations of queen conch and Caribbean spiny lobster have closed seasons that offer some protection to mating and spawning adults. However, their closed seasons do not overlap by design because of the commercial importance of both species but could be modified for increased protection of reproductive activities. The ban on SCUBA and surface-supplied air to the exclusivity of breath-hold diving has certainly been instrumental for protection but only for the small part of the adult conch and lobster populations that cannot be regularly accessed due to depth > 25 m and rough conditions outside lagoons (Tewfik et al. 2019, Tewfik et al. 2020). In addition, while both species have legal minimum sizes (conch shell > 178 mm, lobster tail mass > 128 g), these are significantly smaller than the size of maturity, allowing many juveniles to be harvested legally (Gibson et al. 1983, Wade et al. 1999, Foley and Takahashi 2017, Tewfik et al. 2019, Tewfik et al. 2020). In the case of conch, the application of the small shell length limit since the late 1970s (Gibson et al. 1983), when lip thickness is the only reliable external feature of maturity, has actually reduced the shell length of lipped conch in several fishing grounds, thus compromising fecundity and the size of harvested meat (Fig. 14) (Tewfik et al. 2019, Tewfik et al. 2021). Lobster landings from three fishing grounds all indicate overfishing based on high fishing to natural mortality ratios (F/M) and low spawning potential ratios (SPR) as well as declining tail weights (Tewfik et al. 2020). The mismatch between revised minimum carapace length (> 82.5 mm) and tail mass (> 128 g) for lobsters still allows many immature lobsters to be harvested with incumbent losses in fecundity and meat mass (Wade et al. 1999, Tewfik et al. 2020, Anonymous 2021).

The 2020 banning of fairly inexpensive and widely deployed gillnets will certainly reduce targeted catch and bycatch of larger species including snappers, barracuda and sharks, the later of which were heavily impacted by the introduction of the gear in the 1970s (Graham 2007). (Tewfik et al. 2022). At the same time the catchability of smaller, highly productive species including mojarras and grunts will likely decline and negatively impact subsistence-based food security and the traditional mojarra fishery in northern Belize. This may require area specific or seasonal exceptions for gillnets in addition to broader implementation of minimum sizes for other large species, currently limited to Nassau grouper, to facilitate release when possible. The banning of selected fish families including all angelfish (Pomacanthidae, 0.2% of catch) and all triggerfish (Balistidae, 0.3% of the catch) may be justifiable on ecological grounds (see below), but their impacts for fisheries management are questionable given their limited cultural appeal and presence in local markets or exports (Heyman and Graham 2000, Greers et al. 2020, Tewfik et al. 2022). These family-level protections are in place despite the fact that many threatened larger species of high conservation concern remain legal targets with no minimum sizes or seasons, including goliath and black grouper, cubera and mutton snapper, Caribbean reef, blacktip and both great and scalloped hammerhead sharks (Burgess and Branstetter 2009, Sadovy de Mitcheson et al. 2013, Lindeman et al. 2016b, Lindeman et al. 2016a, Sadovy et al. 2018, Jabado 2024).

The banning of herbivorous fish (Scaridae, Acanthuridae) extraction in Belize since 2009 was motivated to address the coral-algae phase shift by increasing abundance of parrotfish and associated rates of herbivory (McClanahan and Muthiga 2020) and observations that some parrotfish species were being targeted likely in response to more traditional large grouper and snapper species becoming rare (Mumby et al. 2012). However, the response of reef habitat, even within MPAs, to this “herbivore” ban in Belize has been difficult to observe, as elsewhere, with several studies indicating a disconnect between potential hard coral recovery and higher densities of herbivorous fish (Fig. 5) (Suchley et al. 2016, Arias-González et al. 2017, Cox et al. 2017, McClanahan and Muthiga 2020, McField et al. 2024a).

A government-stated 2020 mandate to create fisheries management plans for all commercial species has been implemented extremely slowly (Anonymous 2020). This is especially concerning given that many species traditionally harvested and preferred are threatened groupers (e.g., goliath, Nassau) and snappers (cubera, mutton) that are well documented to be overfished and increasingly rare in visual surveys and catch (Greers et al. 2020, Tewfik et al. 2022, Palomares et al. 2023, McField et al. 2024a). The use (e.g., Nassau grouper) or revision (e.g., queen conch and Caribbean spiny lobster) of biologically sound minimum and possibly maximum sizes to protect juveniles and mega-spawners (Froese 2004, Hixon et al. 2014, Foley and Takahashi 2017, Gnanalingam and Butler 2017, Tewfik et al. 2020) would add a significant layer of sustainable management actions in conjunction with Belize’s extensive protected areas system as would a concerted effort to address environmental impacts of land-based nutrient pollution that threaten hard corals (i.e. coral-algae phase shift) and reef habitat complex integrity (Lapointe 1997, Gibson et al. 2004, Lapointe et al. 2010, Tewfik et al. 2017, Lapointe et al. 2021). The use of species-specific national size limits, focused on maintaining recruitment, may also simplify enforcement by dispensing with the complexities of time and area restrictions for many species. The creation of a comprehensive set of species-specific management plans adopted into law in addition to strengthening of closed area and seasonal enforcement actions would provide the most holistic and sustainable management of fisheries resources and protection of underlying marine ecosystems for the future of all Belizeans and those that care deeply about its people and environment.

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Conflict of Intertest

The authors have declared that no conflict of interests exist.

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