The IUCN Red List of Threatened Species™
ISSN 2307-8235 (online)
IUCN 2008: T195409A46957794
Scope: Global
Language: English

Epinephelus itajara, Atlantic Goliath Grouper

Assessment by: Bertoncini, A.A., Aguilar-Perera, A., Barreiros, J., Craig, M.T., Ferreira, B. & Koenig, C.

View on www.iucnredlist.org

Citation: Bertoncini, A.A., Aguilar-Perera, A., Barreiros, J., Craig, M.T., Ferreira, B. & Koenig, C. 2018. *Epinephelus itajara*. The IUCN Red List of Threatened Species 2018: e.T195409A46957794. http://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T195409A46957794.en

Taxonomy

Kingdom	Phylum	Class	Order	Family
Animalia	Chordata	Actinopterygii	Perciformes	Epinephelidae

Taxon Name: *Epinephelus itajara* (Lichtenstein, 1822)

Synonym(s):

- *Promicrops ditobo* Roux & Collignon, 1954
- *Promicrops esonue* Ehrenbaum, 1915
- *Promicrops itaiara* (Lichtenstein, 1822)
- *Serranus galeus* Müller & Troschel, 1848
- *Serranus guasa* Poey, 1860
- *Serranus itajara* Lichtenstein, 1822

Regional Assessments:

- Gulf of Mexico

Common Name(s):

- English: Atlantic Goliath Grouper, Goliath Grouper
- French: Mérou, Mérou Géant, Têtard
- Spanish: Cherna, Cherne, Guasa, Guato, Guaza, Mero, Mero Batata, Mero Guasa, Mero Güasa, Mero Pintado, Mero Sapo

Taxonomic Source(s):

Eschmeyer, W.N., Fricke, R. and Van der Laan, R. (eds). 2018. Catalog of Fishes: genera, species, references. Updated 04 September 2018. Available at: http://researcharchive.calacademy.org/research/ichthyology/catalog/fishcatmain.asp. (Accessed: 04 September 2018).

Taxonomic Notes:

In 2009, Craig *et al*. confirmed that the Pacific "subpopulations" of *Epinephelus itajara* as a distinct species: *E. quinquefasciatus* (Craig *et al.* 2009).

Assessment Information

Red List Category & Criteria: Vulnerable A2bcd ver 3.1

Year Published: 2018

Date Assessed: November 20, 2016

Justification:

This widely distributed, large-bodied species inhabits hard reef structure and mangrove areas. It is heavily targeted by fishers throughout its range, and has experienced historical population declines as a result. A variety of intrinsic characteristics (e.g., late-maturing, long-lived, aggregate spawning behaviour, predictable occurrence and lack of fear of human presence) make it particularly susceptible to overfishing. It may also be a protogynous hermaphrodite, but that is awaiting confirmation.

In the southeastern U.S., this species experienced a two generation length time period of severe decline from the 1950s to the early 1990s, during which the population declined to near-zero or by at least 84%. A stock assessment model published in 2016 indicated there has been an absolute population reduction of about 33% from 1950 to 2014. A fishing moratorium has been in place in U.S. waters for the past 27 years or since 1990, and the population has been mostly increasing as a result, but is not yet fully recovered. In Brazil, declines in population, including aggregation size, have occurred, but the time period is unknown and the percent is not quantified. A fishing ban has been in place in Brazilian waters since 2002, however, due to lack of enforcement in most areas, poaching continues and fishing mortality reduction is still limited; therefore, the status of its population in Brazil is unknown, but likely remains at a reduced level. Unquantified, but serious declines have also occurred in Cuba, Mexico and Belize. Overfishing and large-scale declines are also likely occurring elsewhere in its range.

In addition to fishing, the pervasive removal and/or degradation of mangroves across its range is a major threat to juvenile survival. Other threats also include reduced genetic diversity, health stresses caused by high mercury concentrations and localised recruitment failures caused by extreme red tide and cold water events given the nearshore and shallow depths often occupied by this species.

Outside of U.S. waters, conservation measures are not sufficient to allow recovery or to prevent continued declines. Even though official historical quantitative records are poor to non-existent, it is clear from multiple well-documented anecdotal observations that a rapid decline occurred over at least the past 50 years as fishing intensity increased and as mangroves declined over the 1970s to 1990s. Therefore, this species is suspected to have declined on a global-level by at least 30% or more since the early 1950s, which covers a time period of about three generation lengths (at least 64.5 years), and is listed as Vulnerable A2bcd.

Improvements in fisheries monitoring and management are needed outside U.S waters, and the closure of the U.S. fishery should remain in place, especially since we suspect that at least an 80% population reduction could occur within the next three generations should the current management be removed. Spawning aggregation sites and mangrove preservation/restoration should be priorities for conservation. The change in status from the previous assessment reflects an improved application of the IUCN Red List Categories and Criteria, as well as a better understanding of available data.

Previously Published Red List Assessments

2011 – Critically Endangered (CR)
http://dx.doi.org/10.2305/IUCN.UK.2011-2.RLTS.T195409A8961414.en

Geographic Range

Range Description:

This species is distributed in the Atlantic Ocean in the west from northeastern Florida, south along the U.S., throughout the Gulf of Mexico and Caribbean Sea, and along South America to Santa Catarina, Brazil (Hostim-Silva et al. 2005) and in the east along West Africa from Senegal to Cabinda, Angola (Friedlander *et al*. 2014, Wirtz *et al*. 2014).

Country Occurrence:

Native: Angola; Anguilla; Antigua and Barbuda; Aruba; Bahamas; Barbados; Belize; Benin; Bonaire, Sint Eustatius and Saba (Saba, Sint Eustatius); Brazil; Cameroon; Cayman Islands; Colombia; Congo; Costa Rica; Côte d'Ivoire; Cuba; Curaçao; Dominica; Dominican Republic; Equatorial Guinea (Annobón, Equatorial Guinea (mainland)); French Guiana; Gabon; Gambia; Ghana; Grenada; Guadeloupe; Guatemala; Guinea; Guinea-Bissau; Guyana; Haiti; Honduras; Jamaica; Liberia; Martinique; Mauritania; Mexico; Montserrat; Nicaragua; Nigeria; Panama; Puerto Rico; Saint Barthélemy; Saint Kitts and Nevis; Saint Lucia; Saint Martin (French part); Saint Vincent and the Grenadines; Senegal; Sierra Leone; Sint Maarten (Dutch part); Suriname; Togo; Trinidad and Tobago; Turks and Caicos Islands; United States; Venezuela, Bolivarian Republic of; Virgin Islands, British; Virgin Islands, U.S.

FAO Marine Fishing Areas:

Native: Atlantic - eastern central, Atlantic - southwest, Atlantic - western central

Distribution Map

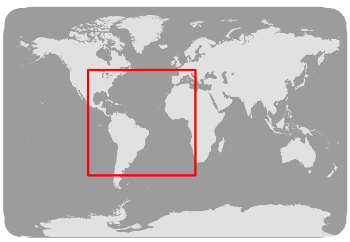
Epinephelus itajara

Range

Extant (resident)

Compiled by:

IUCN Grouper and Wrasse Specialist Group



The boundaries and names shown and the designations used on this map do not imply any official endorsement, acceptance or opinion by IUCN.

Population

This species has undergone a severe population decline throughout its entire range and is now rare where it was formerly abundant due to overfishing (Sadovy and Eklund 1999). In most areas, there has been no indication of population recovery. Unfortunately, there are very few historical data sets that are available to gauge population sizes prior to widespread exploitation (beginning around the 1950s) to serve as a reference for the degree of decline in population sizes. As of the writing of this assessment, the southeastern U.S. is the only area in which an appreciable increase in the population of this species has occurred, this following a moratorium on catch in 1990 (Figure 1 in the Supplementary Information). This is also the only area where reasonably accurate historical landings and effort data are available, and the only area in which a quantitative stock assessment analysis has been completed (SEDAR 47 2016). Even though official historical quantitative records are poor to non-existent, it is clear from well-documented anecdotal observations that a rapid decline occurred over at least the past 50 years as fishing intensity increased and as mangroves declined over the 1970s to 1990s. Therefore, this species is suspected to have declined on a global-level by at least 30% or more since the early 1950s, which covers a time period of about three generation lengths (at least 64.5 years).

There is empirical evidence that the Gulf of Mexico and Atlantic populations should be treated as one unit. Craig *et al*. (2009) found that genetic differentiation occurs between Belize and southwest Florida, and Brazil and Florida. Population genetics studies are currently underway in Florida, and preliminary results indicate that inbreeding is likely due to the moderately low genetically effective population size, which is further evidence for a population crash (C. Koenig pers. comm. 2018).

United States: Based on historical catches by state, the highest abundance of this species in U.S. waters is in the eastern Gulf of Mexico and its historical centre of abundance has been in the Ten Thousand Islands of southwest Florida (Sadovy and Eklund 1999). This species has been targeted by fishers in U.S. waters since colonial times (late 1800s) and fishing mortality was excessively exceeded from 1963-1989, which resulted in severe population declines. In the 1950s and 1960s, this species was common off southern Florida, but by the 1970s and 1980s, the population had been severely depleted to the point that fishers were forced to travel offshore and target spawning aggregations to sustain catch (C. Koenig pers. comm. 2018). Overall, the steepest period of decline in this species is estimated to have begun in the 1950s and extended at least through the early 1990s, which is a 41-year period (1.9 generations), and during this time, the population declined by nearly 84%. This rapid decline in numbers reflects not only a trend in greater harvest, but also intrinsic biological characteristics which make this species particularly vulnerable to over harvest. These characteristics are primarily reflected in their aggregate spawning behaviour (in which several hundred individuals gather in a predicable place and time), which can lead to hyperstability of populations (SEDAR47 2016). Adults were generally uncommon to rare in the mid 1990s, but the population has been recovering upon the implementation of a fishing moratorium in 1990 and continues to the present. The juvenile abundance index in southwest Florida substantially declined during the late 1970s and early 1980s, but has been steadily increasing since the 1990 moratorium until at least 2006 (Cass-Calay and Schmidt 2009). Abundance increased in the mid-1990s directly offshore of the high-quality mangrove nursery of the Ten Thousand Islands and then continued to expand north and south, eventually increasing off Florida's central east coast as well until at least 2007 (Koenig *et al*. 2011, SEDAR47 2016). Abundance declined after 2007 until 2011, and despite some upward trends, abundance remains low overall. A severe red tide event on the West Florida shelf in 2005 and cold-kills in 2008 and 2010 in South Florida estuaries are thought to be the drivers of these declines (SEDAR47 2016). Porch *et al*. (2003) estimated there was a 90-100% chance for a 50% population recovery by 2006-2011. Fisher compliance to the moratorium, however, was found to be lower than 90%, which led to a prediction of less than a 40% chance that the population would recover to the 50% population recovery benchmark by 2020 (Porch *et al*. 2006). Under the moratorium, the estimated relative biomass increased from 0.22 to 0.72 from 1993-2003 and it was categorized as overfished in south Florida in SEDAR6 (2004), but the authors could not determine if overfishing was still occurring due to the lack of data. Considering its high vulnerability to overfishing, a continuation of the moratorium was recommended until further research could determine a complete rebuilding to the target biomass (SEDAR6 2004). From 1993 to 2010 (17 years), relative abundance increased by about five fold, but its overfished status in the South Atlantic could not be determined due to the lack of natural mortality parameters and reliable estimates of moratorium effectiveness (SEDAR 2011). The most recent spawning stock biomass models suggest it is no longer overfished nor experiencing overfishing (SEDAR47 2016); however, the population continues to be skewed towards younger fish 26 years after closure, and will require more time to rebuild the older age classes (Koenig and Coleman 2016).

Stock assessments conducted by the SouthEast Data, Assessment, and Review (SEDAR) have been repeatedly rejected under peer review because of poor historical records and the resulting inability to determine population recovery in terms of size and geographical extent. Despite the large degree of uncertainty in the historical data upon which the most recent stock assessment, SEDAR47 (2016), is based, it is the only available stock assessment with which historical trends in abundance may be gauged. And, while we acknowledge the large degree of uncertainty in this analysis, we posit that the overall trends in the magnitudes of changes are likely to be real even though the absolute values may change with improved historical input data. It is also important to note that the values presented below are based on computer simulations (models) and do not represent direct counts of individuals. We caution that the absolute numbers cited below be interpreted with caution. It should also be noted that this analysis is only for the population of this species in the southeastern U.S. SEDAR 47 (2016) utilized a stock assessment model to hind cast various metrics of abundance to the year 1950. That analysis showed that the average number of individuals among model iterations in 1950 was 513,072 in the southeast U.S. This number remained relatively constant until about 1975, when it declined from a mean of 378,164 individuals in 1975 to a mean of 82,900 individuals in 1991, which represents about an 80% decline over 16 years. After 1991, the number of individuals increased to a high of 1,186,100 in 2006, but declined thereafter until 2011 with a mean number of fish of 298,500, which is a 75% decline over five years. In 2014 (the terminal year of the analysis), there was a mean of 345,700 individuals. From 1950 to the historical low (1991), there was an absolute population reduction of about 84% (over 41 years). From the historical low in 1991 to 2014 there has been a population increase of about 86%. Over the entire period modelled from 1950 to 2014 (the terminal year of the analysis) there has been an absolute population reduction of about 33%. The increases in the numbers of individuals were largely driven by an increase in age-0 fish (which cannot reproduce). During the period 1950-2014, age-0 Atlantic goliath grouper represented 50-85% of the entire population. In 2014 (the terminal year of the analysis), the percentage of age-0 fish was 69%.

Cuba, Mexico and Belize: No fishery assessments have been conducted for this species in the southern Gulf of Mexico and Mexican fishery statistics do not include fishing effort (CONAPESCA). In Mexico, exploitation is unregulated and its population status is poorly known. Some fishery dependent data indicate declining catch trends of this species (Salas *et al*. 2006). Fisher interviews and records from fishing cooperatives off the northern Yucatan indicate that catch has severely declined over about the past 35 years (Aguilar-Perera *et al*. 2009; Figure 2 in the Supplementary Information). In Cuba, lobster fishermen frequently captured this grouper species (R. Claro pers. comm. 2014). Although national statistics are not available, population declines observed in Cuba are at least similar to those observed in Mexico. In Belize, it is considered overfished and undergoing overfishing (Graham *et al*. 2009).

Brazil: Due to a fishing moratorium for this species in place in Brazil since 2002, catch declined by 70% after 2002, but an average of 393 tons per year were poached between 2003-2011 (Giglio *et al*. 2014b). For example, in the state of Maranhão, fishermen have illegally caught large numbers of adults in spawning aggregations (M.O. Freitas pers. comm. 2016). In northern Brazil and on the Amazonian coast, catches are reported to continue and poachers have targeted aggregations and the species is openly captured and traded in local markets (B. Bentes pers. comm. 2016). In Santa Catarina, Brazil, at least two aggregation sites were mapped and aggregation size ranged from two to 60 individuals (Gerhardinger *et al*. 2009), but subsequent dives showed that these aggregations were depleted or had moved (Bueno *et al*. 2016). This species is relatively rare in offshore oceanic islands (Bertoncini *et al*. 2014). Fishers on the Abrolhos Bank noted that the abundance of this species has declined by more than 40% and that poaching continues to occur (Zapelini *et al*. 2017). Spawning populations in northern Brazil are the main source for the population of juveniles in French Guiana (Artero *et al*. 2015a and b). During Reef Check underwater surveys conducted in Brazil from 2002-2017 in 12 localities and 114 reef sites, individuals of this species were observed only three times, all inside fully protected areas (Reef Check Brazil unpub. data); however, this species can occur recurrently in some very predictable places, particularly shipwrecks. Poaching occurs at these sites despite the protected status of this species in Brazil (B. Padovani-Ferreira and Athila Bertoncini pers. comm. 2018).

West Africa: This species is common in shallow waters along mangrove swamps and river mouths near the rivers of Zaire, Ogoué, Cameroon and tributaries as well as near offshore oil platforms of Cabinda and Gabon (Port Gentil), where it is captured by local artisanal fisheries. It is apparently rare in the Gulf of Guinea islands; one specimen was speared in Sao Tomé in 2009 and two juveniles were observed in 2015 (J. Barreiros pers. comm. 2016). Two specimens were taken via spear in Benguela, southern Angola in 2011 (J. de Sousa pers. comm. 2016). One *ca* 250 kg specimen was observed in a fish market in Libreville, Gabon in 2015 (J. Barreiros pers. comm. 2016).

For further information about this species, see Supplementary Material.

Current Population Trend: Decreasing

Habitat and Ecology (see Appendix for additional information)

Adults are most often associated with offshore rocky reefs, wrecks, artificial reefs and oil platforms. It can occur in coral reef habitat, but is much more abundant on rocky reef (Collins 2009; Koenig *et al*. 2011; Giglio *et al*. 2014a,c; Collins *et al*. 2015). Mangroves are the primary juvenile habitat (Koenig *et al*. 2007); juveniles are also occasionally observed on adjacent inshore structured habitats, such as seagrass, tide pools, shallow rocky areas, jetties and docks (Bullock *et al*. 1992; Sadovy and Eklund 1999; Gerber *et al*. 2005; Artero *et al*. 2015a,b; Lobato *et al*. 2016), but return to 'home' sites in nearby mangroves at low tide (C. Koenig pers. comm. 2018). Juvenile distribution in mangroves depends on local water quality, particularly dissolved oxygen content (>4 ppm) and mid-range salinities (>10 ppt; Koenig *et al*. 2007). Within the mangrove system, undercuts and sites with high structural complexity are particularly favoured by juveniles (Frias-Torres 2006). Individuals spend their first 5-6 years in this habitat and reach a meter in length and about 50 lbs in weight as juveniles and are extremely site-specific (C. Koenig pers. comm. 2018). Juveniles move from nearshore shallow habitats at a total length of about one meter to take up residence on offshore reefs (Koenig *et al*. 2007, Collins *et al.* 2015). In French Guiana, the population of this species is comprised primarily of juveniles that utilize rocky habitats on the windward sides of offshore islands due to the ephemeral nature of the Amazonian-influenced mangroves (Artero *et al*. 2015a,b). Large juveniles and adults on offshore structure display strong site fidelity but move from home sites to spawning aggregation sites during July (Koenig *et al*. 2011, Ellis *et al*. 2013, Collins *et al*. 2015) and remain on spawning sites until mid-October. This species primarily consumes fish, crabs and crustaceans (Sadovy and Eklund 1999, Koenig and Coleman 2009, Artero *et al*. 2015a, Freitas *et al*. 2015, Koenig and Coleman 2016). Maximum total length is 250 cm (Heemstra and Randall 1993).

In the southeastern U.S., spawning aggregations occur at the same localities in relatively shallow (10-50 m) water from August through mid-October (Coleman *et al*. 2002, Koenig *et al*. 2011, Ellis *et al*. 2013, Koenig *et al*. 2016, Koenig and Coleman 2016). Adults have been observed to migrate up to 500 km to spawning sites (Ellis *et al*. 2013). It often spawns near wrecks (Collins *et al*. 2015), but also uses natural sites which it often excavates (Koenig *et al*. 2011). Aggregations are relatively small, usually with fewer than 150 individuals, and there is no evidence that spawning occurs outside of aggregations (Sadovy and Eklund 1999). Aggregations have been detected during the austral summer (December to March), with peaks in January and February in shallow waters (<30 m depth) in at least seven Brazilian states (Bueno *et al*. 2016, Giglio *et al*. 2016). Many spawning aggregation sites have been identified in the U.S. Atlantic and Gulf (Mann *et al*. 2009, Koenig *et al*. 2016), and one has recently been documented in southern Brazil as well (Bueno *et al*. 2016).

This species may have a reproductive strategy of diandric protogyny (Koenig and Coleman 2016), but functional hermaphroditism has not yet been confirmed. The total length at first maturation for females was estimated at 105.6 cm and 100% of individuals are mature at 126 cm and age at first maturity is about six (Sadovy and Eklund 1999) or seven years (Bullock *et al*. 1992, Koenig *et al*. 2007, Koenig and Coleman 2016). Growth rates for male and female are similar, averaging >10 cm per year through age six, then slowing to about three cm per year by age 15, and finally declining to less than one cm per year after age 25 (Bullock *et al*. 1992). From individuals sampled between 1977-1990 in the Gulf of Mexico, Bullock *et al*. (1992) recorded that 46% were ≤12 years old, ~78% were ≤ 18 years old, and ~22% of the fish were relatively old (>18 years, up to a maximum age of 37 years). However, it is very likely that this species reaches older ages than 37 years since the specimens from Bullock *et al*. (1992) were collected after this species was already historically overfished and other comparable grouper species are known to reach ages older than this (e.g., *Hyporthodus flavolimbatus* to 85 years and *Mycteroperca interstitialis* to 41 years; SEDAR47 2016). Individuals sampled from 2011 to 2015 on the east coast of Florida, were still ≤12 years old (85%), with 99% of the fish ≤18 years of age. In Brazil, the maximum observed age was 33 years, but most of analysed specimens did not reach 25 years (B.P. Ferreira pers. comm. 2016). In French Guiana, individuals ranged from 1 to 17 years of age, which indicated that the population is probably composed mostly of juveniles (Artero *et al*. 2015b). Natural mortality is 0.18 (SEDAR47 2016), which is supported by estimates of natural mortality through recapture rates (Koenig and Coleman 2016). Further studies on age, growth and reproduction are currently in preparation for publication (C. Koenig pers. comm. 2018).

Based on a longevity of 37 years, age of first maturity of 6 years, and applying the mean generational turnover formula in Depczynski and Bellwood (2006), one generation length is estimated to be 21.5 years.

Systems: Marine

Use and Trade

This species is targeted by commercial and recreational fishers throughout its range. This species is also particularly valuable to dive ecotourism (Shideler and Pierce 2016, A.A. Bertoncini pers. comm. 2016).

Threats (see Appendix for additional information)

Overfishing is a major threat to this species, and its susceptibility to rapid population decline is increased due to heavy exploitation of spawning aggregations (Bullock *et al*. 1992, Sadovy and Eklund 1999). Mangrove coverage, which is responsible for biomass production of this species, has been reduced by at least a third since the 1970s to 1990s, and much more of the habitat is unsuitable as juvenile habitat because of anthropogenic impacts (Sadovy and Eklund 1999, Valiela *et al*. 2001, Koenig *et al*. 2007, C. Koenig pers. comm. 2018). Multiple cohorts were affected by a cold temperature event in 2010, which caused a die-off of juveniles in the Everglades National Park, Florida (Hallac *et al*. 2010). Juveniles have also been impacted by past severe red tide events on the West Florida Shelf (SEDAR47 2016). A study conducted in Florida reported that mean mercury concentrations in individuals of this species were within the range known to cause direct health effects in fish after long-term exposure, and concluded this could cause stress on their populations (Adams and Sonne 2013, Koenig and Coleman 2016). Another study conducted off Belize also found elevated levels of mercury (Evers *et al*. 2009). Ongoing research on the impacts from high mercury concentrations indicate that older males in particular suffer liver damage and/or mortality, and that egg viability is reduced (C. Koenig pers. comm. 2018). Another potential threat to the recovering population is inbreeding, and related genetics studies are currently in preparation for publication (C. Koenig pers. comm. 2018).

Conservation Actions (see Appendix for additional information)

There has been a complete moratorium on harvest of these species in continental U.S. waters since 1990 and in U.S. Caribbean waters since 1993, and it is clear that the implementation of this management measure was the key factor in the increase in individuals in the southeastern U.S. seen at the current time (Porch *et al*. 2003, SEDAR47 2016). The SEDAR47 stock assessment analysis shows that the population can decline by nearly 85% in just 1.9 generations in the absence of this management effort, and therefore, it is highly recommended that this moratorium remain in place (Ferreira *et al*. 2012, Frias-Torres 2012, Koenig and Coleman 2016).

Conservation measures for this species off Mexico are insufficient and improvement in regional and international management, including a formal stock assessment, are needed (Aguilar-Perera *et al*. 2009). Under direction of the IBAMA (Brazilian Environmental Agency), this species has been fully protected in Brazil since 2002. This reduced fishing effort, but did not eliminate it due to insufficient enforcement against poaching (Giglio *et al*. 2014b, B. Padovani-Ferreira and Athila Bertoncini pers. comm. 2018). The inter-Ministry ordinance No.13 of 2015 granted protection for this species until 2023 in Brazilian waters. Spawning aggregation sites should be a priority for conservation. This species has been recorded in several marine protected areas, including the Flower Garden Banks National Marine Sanctuary in the northern Gulf of Mexico (Hickerson *et al*. 2008) and Parque Nacional Guanahacabibes in western Cuba (R. Claro pers. comm. 2014). Research is currently underway on adult and juvenile habitat preference, size, age, movement, aggregation location and behaviour and feeding ecology in U.S. waters. Further research is needed on the genetics and age structure of the offshore portion of the U.S. population to improve stock assessment models (SEDAR47 2016).

Credits

Assessor(s): Bertoncini, A.A., Aguilar-Perera, A., Barreiros, J., Craig, M.T., Ferreira, B. & Koenig, C.

Reviewer(s): Linardich, C.

Contributor(s): Chan, T.T. & Sadovy, Y.

Bibliography

Adams, D.H. and Sonne, C. 2013. Mercury and histopathology of the vulnerable goliath grouper, *Epinephelus itajara*, in U.S. waters: A multi-tissue approach. *Environmental Research* 126: 254-263.

Aguilar-Perera, A., González-Salas, C., Tuz-Sulub, A. and Villegas-Hernández, H. 2009. Fishery of the Goliath grouper, *Epinephelus itajara* (Teleostei: Epinephelidae) based on local ecological knowledge and fishery records in Yucatan, Mexico. *Revista de Biología Tropical* 57(3): 557-566.

Artero, C., Koenig, C.C., Richard, P., Berzins, R., Guillou, G., Bouchon, C. and Lampert, L. 2015a. Ontogenetic dietary and habitat shifts in goliath grouper *Epinephelus itajara* from French Guiana. *Endangered Species Research* 27(2): 155-168.

Artero, C., Murie, D.J., Koenig, C.C., Berzins, R., Bouchon, C. and Lampert, L. 2015b. Age, growth, and mortality of the Atlantic goliath grouper *Epinephelus itajara* in French Guiana. *Endangered Species Research* 28(3): 275-287.

Bueno, L.S., Bertoncini, A.A., Koenig, C.C., Coleman, F.C., Freitas, M.O., Leite, J.R., De Souza, T.F. and Hostim-Silva, M. 2016. Evidence for spawning aggregations of the endangered Atlantic goliath grouper *Epinephelus itajara* in southern Brazil. *Journal of Fish Biology* 89(1): 876-889.

Bullock, L.H., Murphy, M.D., Godcharles, M.F. and Mitchell, M.E. 1992. Age, growth and reproduction of jewfish *Epinephelus itajara* in the eastern Gulf of Mexico. *Fisheries Bulletin* 90: 243-249.

Cass-Calay, S.L. and Schmidt, T.W. 2003. Standardized catch rates of juvenile goliath grouper, *Epinephelus itajara*, from the Everglades National Park Creel Survey, 1973-1999. Sustainable Fisheries Division Contribution SFD-2003-0016.

Coleman, F., Koenig, C. and Eklund, A. 2002. Report to The Curtis and Edith Munson Foundation on the Preliminary Studies Using Pop-up Archival Satellite Tags to Track Movements of Adult Goliath Grouper *Epinephelus itajara* Populations in the Eastern Gulf of Mexico.

Craig, M.T., Graham, R.T., Torres, R.A., Hyde, J.R., Freitas, M.O., Ferreira, B.P., Hostim, M., Gerhardinger, L.C., Andrade, A.B. and Robertson, D.R. 2009. How many species of goliath grouper are there? Cryptic genetic divergence in a threatened marine fish and the resurrection of a geopolitical species. *Endangered Species Research* 7: 167-174.

Depczynski, M. and Bellwood, D.R. 2006. Extremes, plasticity, and invariance in vertebrate life history traits: insights from coral reef fishes. *Ecology* 87: 3119-3127.

Evers, D.C., Graham, R.T., Perkins, C.R., Michener, R. and Divoll, T. 2009. Mercury concentrations in the goliath grouper of Belize: an anthropogenic stressor of concern. *Endangered Species Research* 7: 249-256.

Ferreira, B.P., Hostim-Silva, M., Bertoncini, A.A., Coleman, F.C. and Koenig, C.C. 2012. Atlantic Goliath Grouper – *Epinephelus itajara*. In: Sadovy de Mitcheson, Y. and Colin, P.L. (eds), *Reef Fish Spawning Aggregations: Biology, Research and Management*, pp. 417-422. Springer.

Frias-Torres, S. 2006. Habitat use of juvenile goliath grouper *Epinephelus itajara* in the Florida Keys, USA. *Endangered Species Research* 2: 1-6.

Frias-Torres, S. 2012. Should the Critically Endangered Goliath grouper Epinephelus itajara be culled in Florida? *Oryx* 47(1): 88-95.

Friedlander, A.M., Ballesteros, E., Fay, M. and Sala, E. 2014. Marine communities on oil platforms in Gabon, West Africa: High biodiversity oases in a low biodiversity environment. *PLOS ONE* 9(8): e103709.

Gerber L.R., Heppell, S.S., Ballantyne, F. and Sala, E. 2005. The role of dispersal and demography in determining the efficacy of marine reserve. *Canadian Journal of Fisheries and Aquatic Sciences* 62(4): 863-871.

Graham, R.T., Rhodes, K.L. and Castellanos, D. 2009. Characterization of the goliath grouper *Epinephelus itajara* fishery of southern Belize for conservation planning. *Endangered Species Research* 7: 195-204.

Hallac, D., Kline, J., Sadle, J., Bass, S., Ziegler, T. and Snow, S. 2010. Preliminary effects of the January 2010 cold weather on flora and fauna in Everglades National Park. Biological Resources Branch, South Florida Natural Resources Center, Everglades and Dry Tortugas National Parks, Homestead, FL.

Heemstra, P.C. and Randall, J.E. 1993. *FAO species catalogue. Vol. 16. Groupers of the world (Family Serranidae, Subfamily Epinephelinae). An annotated and illustrated catalogue of the grouper, rockcod, hind, coral grouper and lyretail species known to date*. FAO Fisheries Synopsis. No. 125, Vol. 16, Rome.

Hickerson, E.L., Schmahl, G.P., Robbart, M., Precht, W.F. and Caldow, C. 2008. The State of Coral Reef Ecosystems of the Flower Garden Banks, Stetson Bank, and Other Banks in the Northwestern Gulf of Mexico. NOAA Flower Garden Banks National Marine Sanctuary.

IUCN. 2018. The IUCN Red List of Threatened Species. Version 2018-2. Available at: www.iucnredlist.org. (Accessed: 15 November 2018).

Koenig, C.C. and Coleman, F.C. 2016. Regional age structure, reproductive biology and trophic patterns of adult Goliath Grouper in Florida. NOAA MARFIN Project NA11NMF4330123.

Koenig, C.C., Bueno, L.S., Coleman, F.C., Cusick, J.A., Ellis, R.D., Kingon, K., Locascio, J.V., Malinowski, C., Murie, D.J. and Stallings, C.D. 2016. Diel, lunar, and seasonal spawning patterns of the Atlantic goliath grouper, *Epinephelus itajara*, off Florida, United States. *Bulletin of Marine Science* 93(2): 391-406.

Koenig, C.C., Coleman, F.C. and Kingon, K. 2011. Pattern of recovery of the Goliath Grouper *Epinephelus itajara* population in the Southeastern US. *Bulletin of Marine Science* 87(4): 891-911.

Koenig, C.C., Coleman, F.C., Eklund, A.M., Schull, J. and Ueland, J. 2007. Mangrove as essential nursery habitat for goliath grouper (*Epinephelus itajara*). *Bulletin of Marine Science* 80(3): 567-586.

Mann, D.A., Locascio, J.V., Coleman, F.C. and Koenig, C.C. 2009. Goliath grouper *Epinephelus itajara* sound production and movement patterns on aggregation sites. *Endangered Species Research* 7: 229-236.

McClenachan, L. 2009. Historical declines of goliath grouper populations in South Florida, USA. *Endangered Species Research* 7: 175-181.

Murie, D.J., Parkyn, D.C., Koenig, C.C., Coleman, F.C., Schull, J. and Frias-Torres, S. 2009. Evaluation of finrays as a non-lethal ageing method for protected goliath grouper *Epinephelus itajara*. *Endangered Species Research* 7(3): 213-220.

Pina-Amargos, F. and González-Sansón, G. 2009. Movement patterns of goliath grouper *Epinephelus itajara* around southeast Cuba: implications for conservation. *Endangered Species Research* 7: 243-247.

Porch, C.E. and Scott, G.P. 2001. Rebuilding times for Nassau Grouper and Goliath Grouper. Sustainable Fisheries Division Contribution No. SFD-01/02-171. Southeast Fisheries Science Center, Miami, Florida.

Porch, C.E., Eklund, A. and Scott, G.P. 2006. A catch-free stock assessment model with application to goliath grouper (*Epinephelus itajara*) off southern Florida. *Fishery Bulletin* 104(1): 89-101.

Porch, C.E., Eklund, A.M. and Scott, G.P. 2003. An assessment of rebuilding times for goliath grouper. SEDAR Goliath Grouper Data Workshop Report, March 2003, 11pp.

Sadovy, Y. and Eklund, A.M. 1999. Synopsis of biological data on the Nassau grouper, *Epinephelus striatus* (Bloch, 1792) and the Jewfish, *E. itajara* (Lichtenstein, 1822). NOAA Technical Report NMFS 146, and FAO Fisheries Synopsis 157.

Sala, E., Aburto-Oropeza, O., Paredes, G. and Thompson, G. 2003. Spawning aggregations and reproductive behavior of reef fishes in the Gulf of California. *Bulletin of Marine Science* 72(1): 103-121.

SEDAR. 2011. SEDAR23 Stock Assessment Report: South Atlantic and Gulf of Mexico Goliath Grouper. Charleston, SC.

SEDAR47. 2016. Stock Assessment Report for Goliath Grouper of the South Atlantic and Gulf of Mexico, 2016. In: O.Hop, J. and Munyandorero, J. (eds), FWC Report IHR2016-001. Florida Fish and Wildlife Conservation Commission Fish and Wildlife Research Institute, St. Petersburg, Florida.

Valiela, I., Bowen, J.L. and York, J.K. 2001. Mangrove forests: one of the world's threatened major tropical environments. *BioScience* 51(10): 807-815.

Wirtz, P., Bingeman, J., Bingeman, J., Fricke, R., Hook, T.J. and Young, J. 2014. The fishes of Ascension Island, central Atlantic Ocean–new records and an annotated checklist. *Journal of the Marine Biological Association of the United Kingdom*: 1-16.

Zapelini, C., Giglio, V.J., Carvalho, R.C., Bender, M.G. and Gerhardinger, L.C. 2017. Assessing Fishing Experts' Knowledge to Improve Conservation Strategies for an Endangered Grouper in the Southwestern Atlantic. *Journal of Ethnobiology* 37(3): 478-493.

Citation

Bertoncini, A.A., Aguilar-Perera, A., Barreiros, J., Craig, M.T., Ferreira, B. & Koenig, C. 2018. *Epinephelus itajara*. The IUCN Red List of Threatened Species 2018: e.T195409A46957794. http://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T195409A46957794.en

Disclaimer

External Resources

For Supplementary Material, and for Images and External Links to Additional Information, please see the Red List website.

Appendix

Habitats

(http://www.iucnredlist.org/technical-documents/classification-schemes)

Habitat	Season	Suitability	Major Importance?
9. Marine Neritic -> 9.2. Marine Neritic - Subtidal Rock and Rocky Reefs	Resident	Suitable	Yes
9. Marine Neritic -> 9.10. Marine Neritic - Estuaries	Resident	Suitable	Yes
12. Marine Intertidal -> 12.1. Marine Intertidal - Rocky Shoreline	Resident	Suitable	Yes
12. Marine Intertidal -> 12.6. Marine Intertidal - Tidepools	Resident	Marginal	-
12. Marine Intertidal -> 12.7. Marine Intertidal - Mangrove Submerged Roots	Resident	Suitable	Yes
15. Artificial/Aquatic & Marine -> 15.11. Artificial/Marine - Marine Anthropogenic Structures	Resident	Suitable	Yes

Threats

(http://www.iucnredlist.org/technical-documents/classification-schemes)

Threat	Timing	Scope	Severity	Impact Score
1. Residential & commercial development -> 1.1. Housing & urban areas	Ongoing	Unknown	Unknown	Unknown
	Stresses:	1. Ecosystem stresses -> 1.1. Ecosystem conversion 1. Ecosystem stresses -> 1.2. Ecosystem degradation		
1. Residential & commercial development -> 1.2. Commercial & industrial areas	Ongoing	Unknown	Unknown	Unknown
	Stresses:	1. Ecosystem stresses -> 1.1. Ecosystem conversion 1. Ecosystem stresses -> 1.2. Ecosystem degradation		
11. Climate change & severe weather -> 11.3. Temperature extremes	Ongoing	Unknown	Causing/could cause fluctuations	Unknown
	Stresses:	2. Species Stresses -> 2.1. Species mortality 2. Species Stresses -> 2.2. Species disturbance		
12. Other options -> 12.1. Other threat	Ongoing	Unknown	Unknown	Unknown
	Stresses:	2. Species Stresses -> 2.2. Species disturbance 2. Species Stresses -> 2.3. Indirect species effects -> 2.3.5. Inbreeding 2. Species Stresses -> 2.3. Indirect species effects -> 2.3.7. Reduced reproductive success		
5. Biological resource use -> 5.4. Fishing & harvesting aquatic resources -> 5.4.1. Intentional use: (subsistence/small scale) [harvest]	Ongoing	Whole (>90%)	Rapid declines	High impact: 8
	Stresses:	2. Species Stresses -> 2.1. Species mortality		

5. Biological resource use -> 5.4. Fishing & harvesting aquatic resources -> 5.4.2. Intentional use: (large scale) [harvest]	Ongoing	Unknown	Rapid declines	Unknown
	Stresses:	2. Species Stresses -> 2.1. Species mortality		
5. Biological resource use -> 5.4. Fishing & harvesting aquatic resources -> 5.4.3. Unintentional effects: (subsistence/small scale) [harvest]	Ongoing	Unknown	Unknown	Unknown
	Stresses:	2. Species Stresses -> 2.1. Species mortality		
7. Natural system modifications -> 7.3. Other ecosystem modifications	Ongoing	Majority (50-90%)	Unknown	Unknown
	Stresses:	1. Ecosystem stresses -> 1.1. Ecosystem conversion 1. Ecosystem stresses -> 1.2. Ecosystem degradation		

Conservation Actions in Place

(**http://www.iucnredlist.org/technical-documents/classification-schemes**)

Conservation Actions in Place
In-Place Research, Monitoring and Planning
Action Recovery plan: Yes
Systematic monitoring scheme: Yes
In-Place Land/Water Protection and Management
Conservation sites identified: No
Occur in at least one PA: Yes
In-Place Species Management
Harvest management plan: Yes

Conservation Actions Needed

(**http://www.iucnredlist.org/technical-documents/classification-schemes**)

Conservation Actions Needed
3. Species management -> 3.1. Species management -> 3.1.1. Harvest management

Research Needed

(**http://www.iucnredlist.org/technical-documents/classification-schemes**)

Research Needed
1. Research -> 1.2. Population size, distribution & trends
1. Research -> 1.3. Life history & ecology
2. Conservation Planning -> 2.3. Harvest & Trade Management Plan

Research Needed
3. Monitoring -> 3.1. Population trends

Additional Data Fields

Distribution
Lower depth limit (m): 100
Upper depth limit (m): 1
Habitats and Ecology
Generation Length (years): 21.5

The IUCN Red List Partnership

The IUCN Red List of Threatened Species™ is produced and managed by the IUCN Global Species Programme, the IUCN Species Survival Commission (SSC) and The IUCN Red List Partnership.

The IUCN Red List Partners are: Arizona State University; BirdLife International; Botanic Gardens Conservation International; Conservation International; NatureServe; Royal Botanic Gardens, Kew; Sapienza University of Rome; Texas A&M University; and Zoological Society of London.