Distribution Map - Giant Clam

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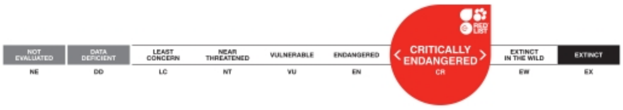
Legend

- 🟧 EXTANT (RESIDENT)
- 🟩 EXTANT & REINTRODUCED (RESIDENT)
- ▨ POSSIBLY EXTANT & REINTRODUCED (RESIDENT)
- 🟪 EXTANT & INTRODUCED (RESIDENT)
- ▨ POSSIBLY EXTINCT
- ▨ POSSIBLY EXTINCT & REINTRODUCED
- ▨ POSSIBLY EXTINCT & INTRODUCED
- 🟥 EXTINCT
- ☰ PRESENCE UNCERTAIN

Compiled by:

Li, R. and Neo, M.L. 2024



Population

Broadly, the paucity of contemporary information on *Tridacna gigas* could reflect the species' rarity on reefs today and the local extinctions. Recent survey efforts on this species generally concurred that it is rare throughout its geographic range (Yusuf *et al*. 2009, Gonzales *et al*. 2014, Harahap *et al*. 2018, Purcell *et al*. 2020, Rehm *et al*. 2021, Dolorosa *et al*. 2024), being found in low numbers (i.e., fewer than 10 individuals per subpopulation) or only in a few patches. Moreover, extensive surveys in the Pacific Islands indicated that the presence of this species is sometimes limited to one individual (C.C.C. Wabnitz pers. obs. 2016). An exception is the Great Barrier Reef in Australia, which is the most extensive area within the natural distribution of *T. gigas* that still supports relatively undisturbed and stable populations (Braley 1984, 1986, 2023) and exhibits evidence of natural recruitment (Braley and Muir 1995, Braley 2023). The largest known subpopulation comprises 277 individuals, including 208 mature ones (Braley 2023, data from 2017). Threats to the Australian population could result in a much higher extinction risk of the species. As reviewed by Neo *et al*. (2017), the lowest density reported was 0.2 individuals per hectare at Tarawa Atoll, Republic of Kiribati (Munro 1988), while the highest density reported was at 431.9 individuals per hectare at Michaelmas Reef, Australia (Pearson and Munro 1991). Furthermore, the uncertain presence or local extinction of this species has been reported from 26 of the 31 localities (Neo *et al*. 2017), strongly suggesting that the global population has declined significantly over the decades.

The age of *Tridacna gigas* has been measured at 76 years (Mecha and Dolorosa 2020), or estimated as at least 60 years (Watanabe *et al*. 2004). According to Mingoa-Licuanan and Gomez (2007), this species exhibits male maturity at 42 cm and females at 53 cm, but no age was provided. Generation length is calculated as: Age of first reproduction + [z * (length of the reproductive period)], with a minimum value of z=0.5 reflecting that mortality is highly skewed toward larvae in broadcast spawning species. Under mariculture conditions, male-phase maturity is at 3.5 years (i.e., a mean size of 18.7 cm, Heslinga and Perron, 1983, Heslinga *et al*. 1984) or at 5 years (Syafyudin Yusuf pers. comm. 2024); the mean value of 4 years is considered. This provides: Age of first reproduction + [z * (length of the reproductive period)] 4 + (0.5 * 56) = 32 years (generation length for the male phase). Female-phase maturity is at around 10 years (H.A. Roa-Quiaoit pers. comm. 2024), so: 10 + (0.5 * 50) = 35 years (generation length for the female phase). Mean value = 33.5 years. This assessment considers 100 years as representing three generations for this species.

Recent research on the population genetics of this species is sparse, but studies from the 1990s indicate discrepancies between current ocean currents and the observed genetic differences among populations in the Central and West Pacific groups, the Great Barrier Reef, and the Solomon Islands (Benzie and Williams 1992, 1995). This discrepancy suggests that despite the large size of *T. gigas*, its dispersal capabilities might be more limited than those of its relatives, potentially increasing its risk of local extinction due to reduced genetic exchange and diversity.

While the knowledge on the mariculture of this species is well-established and the raising of cultured individuals has been successful, the outcomes of translocating these cultured clams for conservation are not well understood or under reported (i.e., it is unclear whether efforts have led to an increase in stock numbers). The exception is restocking efforts in the Philippines that have yielded success in recruitment from their cultured *T. gigas* clams (Gomez and Mingoa-Licuanan 2006, Cabaitan and Conaco 2017, Requilme *et al*. 2021).

Current Population Trend: Decreasing

Habitat and Ecology (see Appendix for additional information)

This species is usually free-living on either sand or hard reef substrata and typically lives in coral reefs with good light penetration. All species of giant clams are known to be simultaneous hermaphrodites. *Tridacna gigas* has one of the highest fecundity rates, producing up to 500 million eggs in a single spawning event. The reproductive periodicity of the Great Barrier Reef population of this species showed diel periodicity to spawning, generally coinciding with incoming tides near the second (full), third, and fourth (new) quarters of the moon phase, occurring over the austral summer (Braley 1984). The mariculture of *T. gigas* is well known and published (Crawford *et al*. 1986; Braley *et al*. 1988), and the reproductive potential is known to some extent. A piece of indirect evidence for the natural reproductive potential of this species is the successful recruitment of juveniles in the Philippines arising from restocked cultured individuals transplanted in 1985, which took approximately 30 years (Cabaitan and Conaco 2017).

Tridacna gigas can produce beneficial outcomes for coral reef ecosystems. For instance, a population of cultured *T. gigas* can produce substantial biomass (e.g., 29 tonnes ha^{-1} yr^{-1} of wet tissue biomass, estimated 1,682 kg dry weight ha^{-1} yr^{-1}) (Barker *et al*. 1988). Giant clams filter large quantities of seawater; even a sparse population of mature *T. gigas* (0.04 clams m^2) on the Great Barrier Reef is capable of filtering over 28,000 L ha^{-1} h^{-1} (Neo *et al*. 2015). An experimental study found that *T. gigas* introduced onto degraded reefs significantly improved fish diversity and abundance compared to control plots with no clams (Cabaitan *et al*. 2008) and their shells provided additional shelter to surrounding organisms (Cabaitan *et al*. 2018, de Guzman *et al*. 2023). An opportunistic observation also discovered that the *T. gigas* gametes were a source of food for the blue sprat (*Spratelloides delicatulus*) (Maboloc and Mingoa-Licuanan 2011). Furthermore, this species is a known host of cyclopoid copepod species (*Anthessius alatus* and *Lichomolgus tridacnae*), pea crabs (*Tridacnatheres whitei* and *Xanthasia murigera*), and pontoniinid shrimps (*Anchistus australis*, *Anchistus demani*, *Anchistus miersi*, *Marygrande mirabilis*, and *Paranchistus armatus*) (Neo *et al*. 2015).

Systems: Marine

Use and Trade

In the past, *T. gigas* was highly regarded as a primary species of interest for food and its shells because of the large body sizes. It is traditionally used as a reserve food when times are difficult (Firdausy and Tisdell 1992, Rehm *et al*. 2021). In contemporary times, this species (lumped together with other giant clams) continues to be harvested for subsistence purposes to varying extents (i.e. opportunistic to intensive harvesting) across its range (Purcell *et al*. 2020, Eurich *et al*. 2023).

In the 1990s, the shells of this species were exported internationally for sale (to Japan, the US, and Europe) (Tisdell 1994). In the Philippines, the large *T. gigas* shells were made into bird baths, wash basins, and garden ornaments (Juinio *et al*. 1987). In Palau, the then Micronesian Mariculture Demonstration Center (MMDC; now known as PMDC) sold thousands of giant clam shells as ornaments in the early 1990s (Heslinga 1996). This species was particularly fashioned as large bowls and soap dishes (Heslinga 1996). In contemporary times, *T. gigas* shells are heavily used in the handicraft carving industry located in Hainan Island, China (Larson 2016). One of the largest *Tridacna* shell markets today is

China. Many of the local fishermen from Tanmen, Hainan, have converted from traditional fishing to the more lucrative tridacnine fishing as their main livelihood (Zhang 2014). Shells of giant clams may have become a substitute for ivory, the import of which is now regulated strictly (Gomez 2015, Larson 2016).

Another specific use/trade of *T. gigas* shells is the floor-tile industry. The shells of this species (lumped together with other giant clams) were previously collected for the highly popular floor-tile industry in Central Java, Jakarta, and East Java (Firdausy and Tisdell 1992). Approximately 10–20 tonnes of processed clam shells were sought every two weeks for the floor-tile industry in Jakarta. *Tridacna gigas* was listed as one of the species sold in the market.

Cultivated *T. gigas* is marketed in the aquarium trade (Mies *et al*. 2017, Militz and Southgate 2021, Vogel and Hoeksema 2024) and possibly as food (Militz and Southgate 2021). Elsewhere, such as in Palau, cultivated *T. gigas* are grown and sold for local consumption (M.L. Neo pers. obs. 2023).

Threats (see Appendix for additional information)

Overexploitation of the species, both for consumption and the shell trade, has been reported as a major cause of population declines (Dawson and Philipson 1989, Hviding 1993, Gomez 2015, Purcell *et al*. 2020, Rehm *et al*. 2021). As the sub-adults/adults of *T. gigas* are free-living (i.e., not attached to the substratum) they are particularly vulnerable to over-extraction (Hviding 1993). The extent of fishing can vary depending on the local coastal communities. For instance, *T. gigas* (with other large clam species) is opportunistically taken during fishing trips targeting other marine resources such as fish and lobsters (Purcell *et al*. 2020). On the other hand, in the Republic of Kiribati, all giant clams (including *T. gigas*) are heavily exploited for subsistence purposes (Eurich *et al*. 2023). Present-day populations of *T. gigas* continue to face high levels of exploitation pressure and habitat deterioration (Gomez 2015). This species remains a valuable coastal resource for both domestic and commercial markets, as it is highly favoured for its meat as food and large shells for the ornamental trade (Larson 2016).

Poaching of *T. gigas* (and other large giant clam species) for food was widespread throughout the region. Prior to the 1980s, commercial exports of giant clam adductor muscles to Asian markets and poaching by long-range foreign vessels were responsible for the severe stock reductions occurring in the Indo-Pacific (Dawson and Philipson 1989). Between 2012 and 2017, coastal resource authorities from various countries (Australia, Cambodia, Malaysia, and the Philippines) reported an increase in the number of fishing boats harvesting giant clams illegally (see Neo *et al*. 2017). Large-scale poaching also poses a major and persistent threat to wild populations.

Climate change has been mentioned as a threat to this species. Studies have confirmed that increased irradiance and seawater temperature bleach *T. gigas*. Specifically, Buck *et al*.'s (2002) experiment found that there is a loss of symbionts, reduced chlorophyll content in symbionts, retention of small zooxanthellae in the tissue, and release of ammonium into the water column. Bleaching of *T. gigas* can further reduce the fitness of host clams, resulting in reduced growth, fecundity/reproduction, and survival (Leggat *et al*. 2003, Sayco *et al*. 2023). In extreme cases, mortality of *T. gigas* has been associated with high seawater temperatures in Australia (Norton *et al*. 1995) and the Philippines (Gomez and Mingoa-Licuanan 1998). The early life stages of this species could also become vulnerable to elevated temperatures, where higher water temperatures promote rapid progression through early larval development but result in lower overall survival as a consequence of abnormal development and reduced post-settlement survival (Enricuso *et al*. 2019).

Other threats include anthropogenic influences such as effluent outfalls and agricultural run-offs that could affect calcified reef invertebrates. For example, *T. gigas* supplemented for three months with elevated ammonium and phosphate had significantly enhanced growth rates, but compromised shell formation (i.e., weaker shells) (Belda *et al*. 1993). The survival rates of *T. gigas* larvae were also found to decrease considerably with reduced salinities and the presence of copper (Blidberg 2003, Sayco *et al*. 2019). In juvenile *T. gigas*, reduced salinities showed effects of reduced heterotrophic feeding (Maboloc *et al*. 2014), and reduced rates of respiration and photosynthesis (Maboloc and Villanueva 2017). When cultured in masses, this species may be threatened by ectoparasite pyramidellids (*Turbonilla* spp.) that have been shown to impact the growth, survival, and physiology of juvenile *T. gigas* (Boglio and Lucas 1997). Apart from the various predators (see Neo *et al*. 2015), bacterial disease could cause mortality in juvenile *T. gigas* (Norton *et al*. 1993).

Conservation Actions **(see Appendix for additional information)**

In 1983, *T. gigas* was formally listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). This is unlike the other giant clam species of subfamily Tridacninae, which are listed on the basis of so-called 'look-alike species', i.e., species whose specimens in trade look like those of species listed for conservation reasons (Wells 1997). Thus, CITES regulates the international trade in any of their parts (shells, tissues, alive or dead).

***In situ* protection of stocks**: *T. gigas* has legal protection under the respective wildlife and fisheries laws in the following countries: Australia, China, Taiwan, Indonesia, Malaysia, Myanmar, Philippines, Singapore, New Caledonia, Solomon Islands, and Pitcairn Islands. In contrast, the South Pacific nations (such as Papua New Guinea, FSM, Palau, Marshall Islands, Republic of Kiribati, Tonga, and Vanuatu) protect their wild stocks by introducing restrictions on harvesting wild giant clams (such as using size, weight or bag limits, gear restrictions or permits) or introducing restrictions on individual uses, including recreational, tourism and aquaculture. The levels of enforcement of laws, however, are unclear and underreported.

Stock enhancement through mariculture: During the 1980s, the efforts to re-establish or supplement depleted populations of giant clams (including *T. gigas*) were mainly funded by the Australian Centre for International Agricultural Research (ACIAR) (Davila *et al.* 2017). The focus of the programme was to breed and release hatchery-reared giant clams back to the wild at local and regional scales. *Tridacna gigas* has been extensively cultured and reintroduced (albeit sometimes limited to a couple of individuals in some areas) to Peninsular Malaysia, Sabah, the Philippines, Fiji, Northern Mariana Islands, Vanuatu, and Tonga, as well as introduced to American Samoa, the Cook Islands, Hawaii (USA), and Samoa (Neo *et al*. 2017). Some of the oldest maricultured *T. gigas* individuals were 34 years old (produced at the Palau's Micronesian Mariculture Demonstration Center in 1982), as well as two others around 40–45 years old. All individuals were previously displayed at the Waikiki Aquarium in Honolulu (Carlson 2012, Heslinga 2012). While, in general, there is little information available regarding the outcomes of restocking in these areas, there has been a notable exception in the Philippines. According to Neo *et al*. (2017), for >20 years, the Marine Science Institute, University of the Philippines, has been culturing giant clam species for the restoration of depleted populations in the Philippines. Several species were initially restocked, but later efforts focused on *T. gigas* (Gomez and Mingoa-Licuanan 2006). Recruits of this species were first observed in the vicinity of Bolinao, Pangasinan, where the broodstock was placed (Cabitan and Conaco 2017). Additional reports on *T. gigas* recruitment around the Philippines have been reported by Requilme *et al*. (2021).

Wildlife trade: Most of the *T. gigas* marketed in the trade are cultured, while the wild-sourced clams are traded in very low quantities (Vogel and Hoeksema 2024). Based on the CITES Trade Database, Tonga was a major exporting country for both wild-sourced and cultured *T. gigas* between 2001 and 2010, while the Philippines was a major exporting country for both wild-sourced and cultured individuals between 2011 and 2019 (Vogel and Hoeksema 2024). Notably, the trade numbers for this species were much lower in recent years (2011–2019) compared to 2001–2010, suggesting a reduced demand for the species. Between 2001 and 2019, the import-export of *T. gigas* shells was greater than that of live specimens (Vogel and Hoeksema 2024). In general, it appears that several cultivation programmes are focused on growing giant clams for the global marine aquarium trade (Mies *et al*. 2017).As the largest extant bivalve species, *T. gigas* holds substantial scientific interest. For instance, its significant size and longevity make it an important model for studying bivalve growth patterns and longevity (Li *et al*. 2024). In another example, subfossil *T. gigas* shells have the potential to yield reliable records of past changes in seasonality and ENSO variability, and mean climate conditions (Welsh *et al*. 2011). This makes its conservation not only an ecological priority but also a significant research imperative.

This species has been assessed as a proposed endangered species in a status review for the US Endangered Species Act (NOAA, 2024).

Credits

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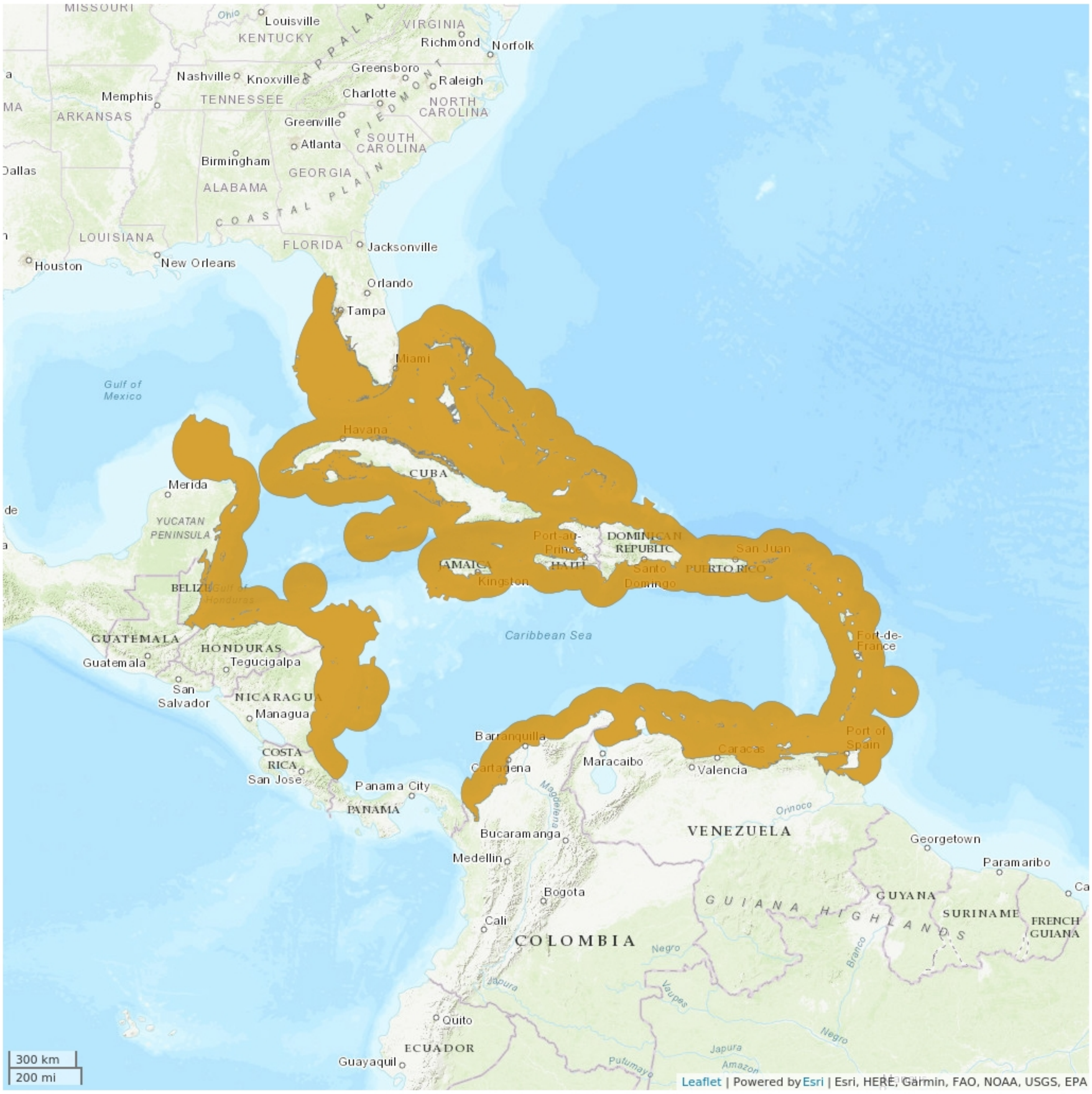
Distribution Map - Stony Coral

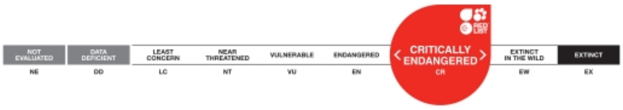


Legend

🟧 EXTANT (RESIDENT)

Compiled by:

IUCN Marine Biodiversity Unit/GMSA 2021



Population

This species is uncommon but conspicuous. It is uncommon or rare in most of its range (Riegl *et al*. 2003, Karpouzli *et al*. 2004, Acosta and Llansó 2004, Ward *et al.* 2006, Bruckner and Bruckner 2006, Quinn and Kojis 2008, García-Sais *et al.* 2008, Hernández-Fernández *et al.* 2008, Clark *et al.* 2009, Scaps and Saunders 2011, Rodríguez-Martínez *et al*. 2012, Sheppard 2013). The species can be locally common in some localities (Geraldes 2003, Kaczmarsky *et al*. 2005, Acosta and Acevedo 2006, Rogers 2009, Bruckner and Hill 2009, Deleveaux *et al.* 2013, Neely *et al.* 2013, Yranzo *et al.* 2014, Marhaver *et al.* 2015, Bernal-Sotelo *et al*. 2019, Cavada-Blanco *et al*. 2020). In the Dominican Republic, it is considered rare but widespread, being reported along the southeast coast and northern reefs, particularly in well-developed reefs located within MPAs (Cortés-Useche *et al*. 2019).

Information on abundance for this species is scarce and scattered in time. The low frequency of occurrence and the distribution of colonies hinder most reef monitoring programs from capturing representative data about this species' abundance (Acosta and Acevedo 2006, Rogers 2009, Brainard *et al.* 2011, Cavada-Blanco *et al.* 2020). Isolated colonies (Szmant 1986, Neely *et al*. 2013, Marhaver *et al.* 2015, Cavada-Blanco *et al*. 2020), as well as abundance of 843 colonies/0.03 km^2 (Bernal-Sotelo *et al.* 2019) and monospecific patches of this species, stretching an area of 98.7 m^2 (Cavada-Blanco *et al.* 2020) has been reported. However, most of the data on abundance is restricted to the Southern Caribbean, the Islands of Providencia and San Andres in Central America and the Florida Keys (Acosta and Acevedo 2006, Neely *et al*. 2013, Marhaver *et al*. 2015, Chan *et al*. 2019, Bernal-Sotelo *et al*. 2019). Though restricted to a few subpopulations within the species’ global range, data indicate a declining trend in the population of this species throughout the Caribbean of at least 80% or greater.

The highest reported abundance for the species corresponds to Archipelago Los Roques National Park (ALRNP), Venezuela (population of 1,794 colonies; Cavada-Blanco *et al*. 2020) in the southern Caribbean and the islands of San Andres and Providencia in the Seaflower Biosphere Reserve (SBR), off the Nicaraguan coast, Colombian Caribbean (846 colonies in 0.03 km^2; Acosta and Acevedo 2006, Bernal-Sotelo *et al.* 2019). In SBR, the number of colonies recorded within reef plots surveyed in 2002 and 2012 increased from 283 to 843 (Bernal-Sotelo *et al*. 2019). However, 94.32% of the increase in abundance was represented by fragments and the authors reported site-level extinctions in 50% of the plots surveyed. The Florida Reef Tract subpopulation is thought to be reproductively extinct with a total population estimated at 610 colonies (Lunz *et al.* 2016 in Chan *et al*. 2019, Neely *et al*. 2021). In the Florida Reef Tract, the number of colonies declined by 93% between 2014 and 2020 (Neely *et al.* 2021). A declining trend off the Yucatan Peninsula due to bleaching and outbreaks of Stony Coral Tissue Loss disease (Alvarez-Filip *et al.* 2019) is also suspected.

In Curaçao, its current abundance (in terms of percentage bottom cover) is 0.09% (SD: 0.13, n=145 sites in 2015 at 10 m depths (Waitt Institute 2017), on Aruba: 0.02% (SD: 0.07, n=53 sites in 2019) (Carmabi unpubl. data). The abundance of this species declined from 0.01% in 1973 to 0.00% in 2005 on Bonaire (de Bakker 2019). Recruitment rates are extremely low and no juveniles were ever observed on Curaçao (Bak and Engel 1979, Vermeij *et al*. 2011). Bonaire and Curaçao have some of the least degraded reefs in the Caribbean (Jackson *et al*. 2014) and the low adult abundance and recruitment failure under favorable conditions indicate that population persistence in this species is likely even more compromised in other locations.

Population growth through sexual recruits is crucial for this species, as it allows the maintenance of genetic diversity and increases fecundity in subpopulations (Marhaver *et al.* 2015). Studies as far back as the 1980s report recruitment of this species as very low (Rogers *et al*. 1984, Szmant 1986). In the past fifteen years, only a few studies have reported evidence of recruitment: Cavada-Blanco *et al*. (2020) reports 22 juvenile colonies in a surveyed area of 6.2 km^2 within Los Roques, Venezuela; Acosta and Acevedo (2006) and Bernal-Sotelo *et al.* (2019) report juvenile colonies (< 15 cm height) in all surveyed sites; and Rogers and Miller (2006) reported a recruit of the species in St. Johns, US Virgin Islands. No evidence of recruitment has been reported for the rest of the species' range, albeit spawning of colonies have been observed in Florida (Neely *et al.* 2013) and Curaçao (Marhaver *et al*. 2015) and gravid colonies in Los Roques (A. Croquer and F. Cavada-Blanco pers. obs. 2015). This has led to hypotheses that the species could be facing a bottle neck (Marhaver *et al.* 2015), as high clonality and low connectivity have also been reported in Florida (Chan *et al.* 2019). In the southern and central Caribbean, subpopulations of the species are dominated by “medium” size colonies (<1 m height; Bernal-Sotelo *et al*. 2019, Cavada-Blanco *et al.* 2020), which might suggest periods of no transition into larger size classes. This could be a strategy to maintain a positive rate of intrinsic population growth in low sexual recruitment (Lirman 2000, Lirman *et al*. 2003, Edmunds and Elahi 2007, Edmunds 2013, Foster *et al.* 2013). However, high rates of fragmentation - as those reported in Florida (Chan *et al.* 2019) and the SBR (Bernal-Sotelo *et al.* 2019) further increase the risk of local extinctions (Marhaver *et al.* 2015).

Current Population Trend: Decreasing

Habitat and Ecology (see Appendix for additional information)

Colonies of this species occur on windward and leeward reef habitats, including fringing reefs, barriers, reef patches and even in spur and grooves, that vary in the type of substrate (pavement, dead coral and sand) and the dominant benthic species (*Acropora cervicornis*, *A. palmata*, *Orbicella annularis*, *Agaricia* spp., *Pseudodiploria* spp., *Madracis* spp., macroalgae and octocorals-antipatharia) (Acosta and Acevedo 2006, Bernal-Sotelo *et al.* 2019, Cavada-Blanco *et al.* 2020, A. Croquer pers. obs. 2015). A colony has been reported near red mangrove roots (Rogers 2017). Higher abundance of colonies seems to occur in reef terraces and patches dominated by octocoral-antipatharia (Bernal-Sotelo *et al.* 2019, Cavada-Blanco *et al*. 2020).

This is the only living species in the genus. It is an EDGE species (Evolutionarily Distinct and Globally Endangered) and is the only one in the Caribbean exhibiting its polyps extended during daylight (Veron 2002, Brown and Bythell 2005). This species exhibits growth plasticity, with encrusting morphotypes in high-energy leeward terraces and massive ones in shallow sandy areas where air-exposure is experienced during low tide (F. Cavada-Blanco and A. Croquer pers. obs. 2015). Most common pillar morphotypes can be easily overturned due to bioerosion at the base; however, these colonies as well as detached pillars or fragments can resume vertical growth and produce new pillars (Geister 1972, F. Cavada-Blanco and A. Croquer pers. obs. 2015). Although the reef area covered by the species is low relative to other shallow reef-building corals, this species contributes to the structural complexity of the habitat (Acosta and Acevedo 2006), creating small reef patches by itself in some instances (Cavada-Blanco *et al.* 2020).This species is dioecious with an annual reproductive cycle (Szmant 1986). The species exhibits both asexual and sexual reproduction (Szmant 1986, Marhaver *et al*. 2015). Spawning occurs 3-5 days after the full moon of August (Neely *et al.* 2013) and September (Marhaver *et al.* 2015). It was thought to be a broadcast spawner, but recent research suggests the reproductive biology of the species is not well known. Evidence from research conducted in Curaçao indicates the species might be sperm-casting with internal fertilisation of eggs inside female colonies after sperm is released by male colonies (Marhaver *et al.* 2015). This further highlights the importance of sex proportion and genetic diversity within close colonies for the species persistence. Szmant (1986) reported a sex ratio of 1:1 for one reef in Puerto Rico, but recent research report imbalanced sex ratios (8.5:1 male:female; Kabay 2016) and hermaphrodite colonies in Florida (Neely *et al*. 2018). The latter has been proposed as a reproductive strategy in small sex imbalanced subpopulations (Neely *et al.* 2018).

The age at first maturity of most reef-building corals is typically three to eight years (Wallace 1999). Based on this, we infer that the average age of mature individuals of this species is greater than eight years. Based on average sizes and growth rates, we also infer that the average length of one generation is 10 years. Longevity is not known, but is likely to be greater than 10 years. Therefore, any population decline rates estimated for the purposes of this Red List assessment are measured over a time period of 30 years.

Systems: Marine

Use and Trade (see Appendix for additional information)

This species is not utilized.

Threats (see Appendix for additional information)

This species is particularly susceptible to bleaching, disease, and other local threats (Aronson *et al*. 2008, Bernal-Sotelo *et al*. 2018, Cavada-Blanco *et al*. 2020). Stony coral tissue loss disease (SCTLD) poses the most immediate threat for this species, due to its high contagion. In Yucatan, Mexico, this species showed the highest prevalence for the disease among 11 reef-building species (Alvarez-Filip *et al.* 2019) and the local subpopulation has been reduced by almost 90% (L. Alvarez-Filip pers. comm. 2020). In Florida, SCTLD has caused a widespread loss in coral cover, affecting large areas (Muller *et al*. 2020) with reports of colony loss of 86% in two years for the species (Kabay 2016). While the prevalence of other diseases (i.e. Black Band, White Plague) in the southern Caribbean and the Seaflower Biosphere Reserve is relatively low, both abundance and partial mortality are related to the condition of the species' habitat (Cavada-Blanco 2018, Bernal-Sotelo *et al*. 2019), increasing the species susceptibility to local extinctions due to habitat degradation. In northern reefs of the Dominican Republic and in several locations of Haiti, extensive mortalities likely associated with SCTLD were observed in 2019 during Reef Check surveys (Ruben Torres pers. comm. 2020).

Caribbean reef-building coral populations have suffered considerable declines over the past 40 years due to the effects of diseases (Alvarez-Filip *et al*. 2011). White plague type 2 (WPD-II) was perhaps the first coral disease to become highly prevalent in populations of this species across the Caribbean. In Los Roques, Venezuela, a WPD-II epizootic event, almost extirpated this species from Madrizquí reef (Croquer *et al*. 2003, 2005). For the past four years, the novel stony coral tissue loss disease (SCTLD) has caused massive die-offs of colonies in Florida (Muller *et al.* 2020) and Mexico (Alvarez-Filip *et al.* 2019), and is now confirmed in the Greater Antilles (Irazabal and Rodriguez 2020). SCTLD has caused an 86% decline in colony abundance in Florida (Kabay 2016). In the Dominican Republic and Haiti, large numbers of dead colonies potentially affected by SCTLD were spotted in 2019, but the disease was not active (Ruben Torres pers. comm. 2020). Similarly, in the Yucatan Peninsula, this species had the highest prevalence among eleven susceptible species of reef-building corals (Alvarez-Filip *et al*. 2019). The

SCTLD can spread between 90 and 100 m per day (Muller *et al*. 2020), potentially decimating entire subpopulations of the species in less than a year. Therefore, the SCTLD currently represents the most important threat to the persistence of this species and other reef-building coral species in the Caribbean.

Localised threats can also drive declines in subpopulations of the species and ultimately lead to local range contractions (Sotelo-Bernal *et al*. 2019, Cavada-Blanco *et al*. 2020). Variability in health status and abundance among reefs within the same MPA in Providencia Island (Colombia) and Los Roques (Venezuela), suggest that overfishing, damaging fishing practices, poorly managed tourism and land-based pollution can reduce colony abundance and deteriorate the species' habitat. With low recruitment and potentially low diversity of genets within subpopulations, recovery from human-caused episodic events and natural disasters such as hurricanes may be slow for the species, and in some instances, cause functional or reproductive extinction (Neely *et al*. 2021).

Global stressors have placed coral reefs at a high risk of collapse (Hughes *et al*. 2017, Franca *et al.* 2020). Increased sea surface temperature as a result of anthropogenic climate change, has also increased the frequency of bleaching events (Hughes *et al*. 2018). The pace at which climate change impacts coral reefs does not allow reef-building corals to recover (Pandolfi *et al*. 2011), causing a rapid decline globally with changes in the composition and structure of coral reefs (Franca *et al.* 2020). Projections using IPCC scenarios suggest that marine heatwaves will increase in frequency and extent, with some scenarios projecting annual severe bleaching to be experienced by all tropical shallow reefs by the end of this century (UNEP 2020). Habitat deterioration and loss by bleaching events can also cause important declines in the species' population. Both abundance and occurrence of colonies of this species are significantly affected by the type of habitat and its condition (Cavada-Blanco *et al*. 2020). Because the species has a low frequency of occurrence, a widespread bleaching event can potentially eliminate local subpopulations.

Conservation Actions **(see Appendix for additional information)**

Conservation of this species is mostly represented by area-based protection through marine protected areas (MPAs) with differing categories of protection across the species global range. However, representativeness of the species is not guaranteed as some MPAs might not include part of the species' national range (Anderson *et al.* 2012) and management effectiveness, severity and scope of local threats vary within and between MPAs (Bustamante *et al*. 2014, Suchley and Alvarez-Filip 2018). Some of the MPAs where the presence of this species has been confirmed include the Florida Keys National Marine Sanctuary (US), Biscayne National Park (US), and Buck Island Reef National Monument (US), the Hol Chan Marine Reserve (Belize), Exuma Cays Land and Sea Park (Bahamas), Seaflower Biosphere Reserve (Colombia), Archipelago Los Roques National Park (Venezuela), Cuare Wildlife Refuge (Venezuela), Boca de Canasi Managed Floristic Reserve, Ciego de Avila National Park (Cuba), Bonaire National Marine Park (Bonaire), La Parguera and Mona and Desecho Island Natural Reserve (Puerto Rico), among others. All corals are listed on CITES Appendix II, which controls their trade to avoid uses that might endangered the species survival. However, adoption of the appendix varies among parties in terms of both legal frameworks and enforcement to regulate extraction and trade. The species has been included within national Red Lists and other schemes used to prioritise threatened species. Some of these include México, Cuba, Dominican Republic, Nicaragua, the Netherlands Antilles and the Endangered Species Act in the U.S., which extends to Puerto Rico and the US Virgin Islands. The threat category assigned to the species varies among countries. Ex-situ conservation measures for reintroduction in restoration projects have started in some parts of the species’ range (A. Croquer pers. comm. 2020). Fundación Dominicana de Estudios Marinos has significantly contributed with assisted sexual propagation for this species. From 2019 to 2020, they have collected eggs and sperm in Bayahibe, they have successfully fertilized to produce millions of larvae for assisted settlement and outplant thousands of spats into restoration sites (Sellares Blasco *et al.* 2021, Villalpando *et al.* 2021). In 2020, FUNDEMAR reported the first ever recorded recruit reared in the laboratory that survived the first 8 month bottleneck in the field (Villalpando *et al.* 2021).

More research in reproductive ecology including other subpopulations of the species’ global range are needed. Recommended measures for conserving this species include: (1) research in reproductive ecology, genetic diversity and connectivity among subpopulations that are representative of the entire global range; (2) research into the epizootiology and dynamics of the SCTLD and potential interventions methods to decrease the disease contagion and colony mortality; (3) updated assessments of the species status within national distribution ranges and the establishment of national monitoring programmes adequate to detect changes on the species status in a timely manner as part of a coordinated regional effort; (4) increase representativeness of the species within absolute reserves and no-take zones throughout its range; (5) improve management efficacy and conservation value of existing MPAs to reduce local threats; (6) integrate land-marine management to reduce land-based pollution; and (7) continue efforts of assisted sexual propagation to restore natural populations with different genotypes.

Credits

Assessor(s):	Cavada-Blanco, F., Croquer, A., Vermeij, M., Goergen, L. & Rodríguez-Martínez, R.
Reviewer(s):	Obura, D.O., Polidoro, B., Gutierrez, L., Kemppinen, K. & Linardich, C.
Authority/Authorities:	IUCN SSC Coral Specialist Group

Distribution Map - Rainbow Parrotfish

Sources: Esri, HERE, DeLorme, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, MapmyIndia, © OpenStreetMap contributors, and the GIS User Community

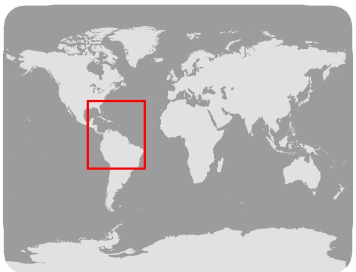
Scarus guacamaia

NE DD LC NT NEAR THREATENED VU EN CR EW EX

Range

🟨 Extant (resident)

Compiled by:
International Union for
Conservation of Nature (IUCN)



Population

Scarus guacamaia has experienced significant historical declines. In most of its range, this species is still fished, but given its current and natural rarity it is not often caught. This species is also threatened throughout its range from the continued loss of coral reef and mangrove habitat from coastal development and extraction. Based on anecdotal information from Glovers Reef in the Caribbean (D. Wesby pers comm 2009), *S. guacamaia* has undergone local extinction in the past 30 years. Schools of this parrotfish were commonly observed in the 1960s when several of the islands had well-developed mangrove habitats (Mumby *et al.* 2004). Within this species' range, aggregated parrotfish catch landings show an increase that began in 2000 with a spike in 2003 and a 27% decline since then (FAO 2012). The pattern in catch landings are concordant to the shift in target preference to herbivorous fishes and planktivores with the removal of the piscivores (Butler *et al.* 1993, Debrot *et al.* 2008, Mumby *et al.* 2012).

Scarus guacamaia is naturally rare, a characteristic that is shared by other large labrids (Choat *et al.* 2006). However, it achieves relatively high densities in protected areas; for example, ~16 individuals/10,000 m² were observed in Bonaire, where it has been protected since 1979 (J.H. Choat pers comm. 2012) and 9.30 ± 3.79 ind/1,000 m² were recorded in Los Roques, where fishing is restricted (Debrot *et al.* 2008). In comparison, densities of *S. guacamaia* and other big-bodied parrotfishes at four localities in the Caribbean open to fishing were 10-fold lower than those within the Los Roques marine reserve, and were absent in Barbados where there was an extensive trap fishery for reef fishes and where parrotfishes are prized components of the reef fishery (Debrot *et al.* 2008). In Las Aves archipelago, an uninhabited area with an uncontrolled fishery targeting piscivorous reef fishes, only 0.4 individuals/10,000 m² were recorded for this species, while 15–16 individuals/10,000 m² were recorded for *Scarus coelestinus*, a large-bodied scarine (J.H. Choat pers comm. 2012). In Bermuda, where parrotfishes have been protected since 1978, this species was observed in big schools (15–30 individuals) (J.H. Choat pers. comm. 2012). Providing further support of the conservation dependency of this species, in a marine reserve in the Bahamas, the Exuma Cays Land and Sea Park, protection has resulted in an increase in total biomass inside the reserves for large-bodied parrotfishes such as *Scarus vetula,* although *S. guacamaia* was not reported in this study (Mumby 2006). This species appears to be uncommon elsewhere in its range (San Blas, Lee Stocking Island, Bahamas, Barbados, Belize, Curacao, St. Croix, St. Thomas, Jamaica and Margarita Island). and at a number of locations only small individuals were seen but none were recorded in counts (J.H. Choat pers comm. 2012).

Current Population Trend: Decreasing

Habitat and Ecology **(see Appendix for additional information)**

Scarus guacamaia is the largest parrotfish in the Atlantic (Dorenbosch *et al.* 2006, Bellwood and Choat 2011) reaching a maximum size of 1.2 m (Choat *et al.* 2006) and a maximum age of 16 years (J.H. Choat pers comm. 2012). This species recruits primarily to mangroves (Mumby *et al.* 2004, Dorenbosch *et al.* 2006, Mumby and Hastings 2008, Bellwood and Choat 2011), and although it is suggested to be functionally dependent on mangroves, juveniles have been collected on rocks in Bermuda and adults have been found on isolated rocky islands 30–50 km from the nearest mangroves off the coast of Venezuela (Robertson pers comm. 2012). This species therefore appears to recruit to a variety of habitats. *Scarus guacamaia* is classified as a detritivore, with detritus/bacterial complexes and meiofauna as the primary food items. It also feeds on sponges (Dunlap and Pawlik 1998) and feeds more similarly to *Sparisoma* spp. particularly *Sparisoma viride*, than to other *Scarus* spp. (Burkepile and Hay 2011).

Systems: Marine

Use and Trade (see Appendix for additional information)

This species is harvested for food and is an important component of subsistence and commercial fisheries in many parts of the Caribbean.

Threats (see Appendix for additional information)

Scarus guacamaia is usually associated with shallow waters and in reefs with extensive sheltered lagoonal/backreef areas (J.H. Choat pers comm. 2012). There are a number of parallels with *Bolbometopon muricatum*, the largest parrotfish in the Indo-Pacific, currently listed as Vulnerable on the IUCN Red List of Threatened Species. Intrinsic life history characteristics such as large size, natural rarity and shallow foraging areas render both species particularly susceptible to overfishing (particularly spearfishing). In addition, both species recruit into very shallow water, sheltered reef and mangrove sites that are increasingly impacted by habitat modification and degradation. Unlike *B. muricatum*, which inhabits a wide oceanic basin and could find refuge on isolated oceanic islands in the Indo-Pacific, *S. guacamaia* inhabits a smaller oceanic habitat and may not have access to the types of refuge available for *B. muricatum.*

More than 30% (37%) of coral reef habitat within the distributional range of *S. guacamaia* has been destroyed or is declining; this coral reef habitat makes up 7% of the species' range (Comeros-Raynal in prep.). In addition, the Caribbean region has the second highest mangrove area loss relative to other global regions with ~24% of mangrove area lost over the past 25 years (FAO 2007, Polidoro *et al.* 2010). The greatest rate of mangrove deforestation is occurring in the Americas (2,251 km²/year) and exceeds that of tropical rainforests (Mumby *et al.* 2004).

Parrotfishes show varying degrees of habitat preference and utilization of coral reef habitats, with some species spending the majority of their life stages on coral reefs, while others primarily utilize seagrass beds, mangroves, algal beds, and /or rocky reefs. Although the majority of the parrotfishes occur in mixed habitat (primarily inhabiting seagrass beds, mangroves, and rocky reefs) approximately 78% of these mixed habitat species are experiencing greater than 30% loss of coral reef area and habitat quality across their distributions. Of those species that occur exclusively in coral reef habitat, more than 80% are experiencing a greater than 30% of coral reef loss and degradation across their distributions. However, more research is needed to understand the long-term effects of habitat loss and degradation on these species populations. Widespread coral reef loss and declining habitat conditions are particularly worrying for species that depend on live coral reefs for food and shelter especially as studies have shown that protection of pristine habitats facilitate the persistence of adult populations in species that have spatially separated adult and juvenile habitats. Furthermore, coral reef loss and declining habitat conditions are particularly worrying for some corallivorous excavating parrotfishes that play major roles in reef dynamics and sedimentation (Comeros-Raynal *et al*. 2012).

Conservation Actions (see Appendix for additional information)

This species is found in a number of marine reserves in parts of its range. Furthermore, in Bermuda all species of parrotfishes are protected under the Fisheries (Protected Species) Order 1978. In addition, there have been recent protective measures put in place for parrotfishes in Belize, wherein the fishing of grazers, defined as any scarinae species and Acanthuridae species, is prohibited. In the Turks and Caicos, the fishing and selling of any species of parrotfish is prohibited, and the Caribbean Management Council, which comprises the Commonwealth of Puerto Rico and United States Virgin Islands, has prohibited the harvest and possession of Midnight Parrotfish (*Scarus coelestinus*), Blue Parrotfish (*Scarus coeruleus*), and *S. guacamaia* and reduced parrotfish harvest in St. Croix. In Bonaire, a recent law (instated in 2010) was passed banning the harvesting of parrotfish (Steneck *et al.* 2011).

Credits

Assessor(s):	Choat, J.H., Feitosa, C., Ferreira, C.E., Gaspar, A.L., Padovani-Ferreira, B. & Rocha, L.A.
Reviewer(s):	Robertson, R., Craig, M.T. & McIlwain, J.
Contributor(s):	Raynal, M.

Distribution Map - Red Mangrove

Sources: Esri, HERE, DeLorme, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, MapmyIndia, © OpenStreetMap contributors, and the GIS User Community

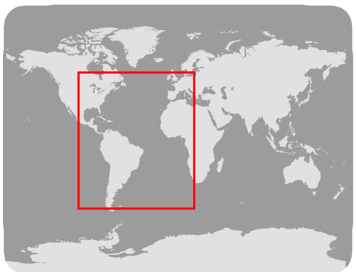
Rhizophora mangle

Range

🟨 Extant (resident)

Compiled by:
International Union for
Conservation of Nature (IUCN)

NE DD **LC** LEAST CONCERN NT VU EN CR EW EX



Population

This species can be common in at least some parts of its range. In Belize for example, it is the most common species, particularly in basin environments along river banks and the coast (Murray *et al*. 2003). It is expanding its range in northern Florida and is characterized as invasive on two Pacific islands.

Specific population information exists for this species in the following areas:

At the mouthouth of Lostman's River in Everglades National Park, FL, U.S., 611 individuals were counted over six transects totaling 0.26 ha (McCoy *et al*. 1996).

In Laguna de Celestun, Yucatan, Mexico, combined data for *A. germinans, R. mangle* and *L. racemosa* show basal area ranges from 21 square meters/ha to 36 square meters/ha (Herrera-Silveira and Ramirez-Ramirez 1998).

In Parque Nacional Morrocoy, Venezuela, data show 68% *R. mangle*, 29% *L. racemosa*, 8% *A. germinans*, 1% unidentified total tree density, and 348 trees/0.1 ha (Bone *et al*. 1998).
Current Population Trend: Decreasing

Habitat and Ecology **(see Appendix for additional information)**

This species typically grows in the intertidal regions of sheltered tropical and subtropical coasts (Saenger 2002). It typically dominates the zone proximal to open water (McKee 1995). It grows as a shrub to small tree from 1-8 m in the Caribbean (Howard 1989) and can be reproductively mature at <1 m (Tomlinson 1986). This species is typically found from 20-35 psu. However, it can occur in brackish estuarine systems as well as in hypersaline conditions (50+ psu) where the growth form is commonly dwarf (1-2 m).

In Port Royal (17°56'N, 76°79'W), *R. mangle* grows in a dense monospecific stands, bordering all wetland water bodies. It is also present in a mixed zone between the *R. mangle* monospecific stands and monospecific stands of *C. erectus*, which is the border between the wetland and terrestrial zones (Alleng 1998).

Flowering occurs annually in mid-winter and spring within the wider Caribbean. Propagule size can be variable, shown to be larger in areas of higher rainfall (Tyagi 2003). Production of viviparous propagules is abundant and maintained on the parent tree for 3-6 months. Once dropped, propagules can subsist for extended periods afloat prior to rooting. Successful growth requires a canopy break or transport to open area to grow to maturity.

Fringing *R. mangle* (in association with seagrass beds) provide critical for Caribbean parotfish (*Scarus guacamaia*), a species listed as vulnerable on the Red Data List. This species is also associated with stabilization of sandy beaches critical for sea turtle nesting habitat, among countless other critical habitat functions.

Systems: Terrestrial, Marine

Use and Trade **(see Appendix for additional information)**

This species has limited direct economic value, but with continued pharmaceutical development, this may change. It is also probably extracted for subsistence fuelwood, charcoal and dyes.

Threats (see Appendix for additional information)

Disease and herbivory can impact *R. mangle* (Farnsworth and Ellison 1991, Brooks and Bell 2002, Sousa *et al*. 2003) but are not likely to pose a serious conservation threat at this time. Also decreases in *R. mangle* have been documented due to land reclamation and clear cutting for marina development (Ellison and Farnsworth 1996, Farnsworth and Ellison 1997, Moore 2006). Although local estimates are uncertain due to differing legislative definitions of what is a 'mangrove' and to the imprecision in determining mangrove area, current consensus estimates of mangrove loss in the last quarter-century report an approximately 17% decline in mangrove areas in countries within this species range since 1980 (FAO 2007).

All mangrove ecosystems occur within mean sea level and high tidal elevations, and have distinct species zonations that are controlled by the elevation of the substrate relative to mean sea level. This is because of associated variation in frequency of elevation, salinity and wave action (Duke *et al*. 1998). With rise in sea-level, the habitat requirements of each species will be disrupted, and species zones will suffer mortality at their present locations and re-establish at higher elevations in areas that were previously landward zones (Ellison 2005). If sea-level rise is a continued trend over this century, then there will be continued mortality and re-establishment of species zones. However, species that are easily dispersed and fast growing/fast producing will cope better than those which are slower growing and slower to reproduce.

In addition, mangrove area is declining globally due to a number of localized threats. The main threat is habitat destruction and removal of mangrove areas. Reasons for removal include cleared for shrimp farms, agriculture, fish ponds, rice production and salt pans, and for the development of urban and industrial areas, road construction, coconut plantations, ports, airports, and tourist resorts. Other threats include pollution from sewage effluents, solid wastes, siltation, oil, and agricultural and urban runoff. Climate change is also thought to be a threat, particularly at the edges of a species range. Natural threats include cyclones, hurricane and tsunamis.

Conservation Actions (see Appendix for additional information)

There are no conservation measures specific to this species, but its range may include some marine and coastal protected areas. Continued monitoring and research is recommended, as well as the inclusion of mangrove areas in marine and coastal protected areas.

Credits

Assessor(s): Ellison, A., Farnsworth, E. & Moore, G.

Reviewer(s): Polidoro, B., Livingstone, S. & Carpenter, K.E.

Distribution Map - Reef Shark

Legend

🟧 EXTANT (RESIDENT)

Compiled by:

IUCN SSC Shark Specialist Group 2020

Population

Genetic information shows residency and/or site-fidelity to specific locations by individual Caribbean Reef Sharks. However, similar to other reef shark studies, the level of genetic connectivity among populations is context-dependent, with sharks occupying isolated habitats showing greater genetic differentiation compared with those sharks occupying semi-isolated or continuous reef habitats. Furthermore, Caribbean Reef Sharks have low genetic diversity and a shallow mitochondrial population history suggesting historical demographic fluctuations (Bernard *et al.* 2017).

In Belize, this species appears to have initially had a stable population up to 2013 based on longline catches (Bond *et al.* 2017). Trend data from Belize and the Bahamas was analysed over three generation lengths using a Bayesian state-space framework (Sherley *et al.* 2020, Winker *et al.* 2020). This analysis yields an annual rate of change, a median change over three generation lengths, and the probability of the most likely IUCN Red List category percent change over three generation lengths (29 years) (see **Supplementary Information 1**). The Belize relative abundance time-series based on baited remote underwater video stations (BRUVs) from 2009 to 2018 (G. Clementi and D. Chapman unpubl. data 2019) indicated that annual abundance declined by 15.4%, consistent with an estimated median reduction of 99.2% over the past three generation lengths (29 years). Longline catch data from the Bahamas from 1979–1984 and 2011–2013 (Talwar *et al.* 2020) suggest that the population has been relatively stable, increasing annually by 0.8% over the past three generation lengths (29 years) (**Supplementary Information 1**). The Bahamas data is two short time-series with a 30-year gap and should be interpreted with caution; the stability and slow increase may be due to an actual population increase but may also reflect a habitat shift from areas of higher human activity to the sampled area that has less human activity (Talwar *et al.* 2020). Population of the Caribbean Reef Shark have remained relatively stable since the 1980s in areas where the commercial exploitation of sharks is prohibited dubbed “shark sanctuaries” (e.g. Bahamas). A comparison of the number of sharks per reef throughout the Caribbean Sea based on BRUVs data also suggest abundance is six times higher in areas where protections are in place (D. Cardenosa and D. Chapman unpubl. data 2019). A preliminary estimate indicates that 20% of the Caribbean Sea is designated as “shark sanctuary” or provides other similar safeguards for Caribbean Reef Shark.

In Caribbean Colombia, this species is common and increasing, but may have undergone some past reductions due to illegal fishing and was thus assessed as Near Threatened nationally (Gomez *et al*. 2017). In Venezuela, there are no data, but this species is caught in high numbers (Tavares 2009), and it is suspected that with a lack of management this has led to declines there. There are also no data from the Guianas, but artisanal fisheries there remain unmanaged and may be a cause for concern for this species. In Brazil, this species is listed as Vulnerable due to the level of exploitation there and notable declines in landings in the State of Maranhão and the Trindade and Martin Vaz archipelago, with a suspected reduction in population size of 30% (P. Charvet unpubl. data 2019). In fact, this species is suspected to have been lost from coastal Brazil and is now likely only found off four islands (Paracel, Noronhua, Abrolhos, Trindade), as the last confirmed record was from Cearà State in 1987 (F.M. Santana pers. comm. 25/04/2018). It was formerly common in places such as São Paulo (Sadovsky 1967).

A total of 4,834 BRUVS deployments at 109 reefs in 18 jurisdictions (of 15 nations) were used in the analysis of depletion and Red List status of Caribbean Reef Shark. In total, Finprint sampled jurisdictions that covered 88.2% of the world’s coral reefs (MacNeil *et al*. 2020). Caribbean Reef Shark depletion estimates were based on jurisdictions that contained 73.6% of the coral reefs within its known range. Depletion was estimated to be greatest in locations such as Jamaica, Dominican Republic, mainland Colombia and the Islands (**Supplementary Information 2**). On coral reefs across the Caribbean, the reef area weighted population depletion level for Caribbean Reef Shark was 52.5% (standard error 40.4–64.5%) (see **Supplementary Information 2**). Assuming that this depletion occurred over the past three generation lengths (29 years), this equated to an estimated population reduction of 52.5% (standard error 40.4–64.5%) meeting Endangered A2b. Error estimates include both the Vulnerable and Endangered Red List categories, but given the majority of the range is within the Endangered category, and the estimate is likely to be an underestimate of the reduction, and due to levels of exploitation, we conclude that the most suitable category for Caribbean Reef Shark is Endangered A2bcd.

For further information about this species, see Supplementary Material.

Current Population Trend: Decreasing

Habitat and Ecology (see Appendix for additional information)

The Caribbean Reef Shark is a tropical inshore shark inhabiting coral reefs on continental and insular shelves across a wide depth range from the surface to a depth of 378 m (Chapman *et al*. 2007, Castro 2011, Weigmann 2016). It reaches a maximum size of 295 cm total length (TL), males mature at 150–170 cm TL and females mature at 180–190 cm TL (Pikitch *et al.* 2005, Ebert *et al.* 2013, Tavares 2009). Reproduction is viviparous with litter sizes of 3–6, an approximately one year gestation and biennial reproductive cycle, and size-at-birth of 70 cm TL (Ebert *et al.* 2013). Generation length was estimated as 9.6 years based on an age-at-maturity of 4.2 years estimated from the reported length-at-maturity and back-transforming length into age using the growth curve and maximum age of 15 year (Tavares 2009).

Systems: Marine

Use and Trade

The Caribbean Reef Shark is used for meat, fins, leather (skin), oil (livers) and fishmeal (from carcasses). In Colombia, the jaws and livers are used for ornaments and oil, respectively, while the meat is only occasionally used as it is not easily marketed. Based on trimmings of shark fins from Guangzhou and Hong Kong, Caribbean Reef Shark were not reported as part of the fin trade (Cardeñosa *et al.* 2020). SCUBA diving with these sharks has become a very popular form of ecotourism. For example, in 2007, divers experienced an estimated 73,000 shark interactions in the Bahamas, generating roughly US $78 million in annual revenue (Cline 2008), and Haas *et al.* (2017) estimated a contribution of US $113 million to the Bahamian economy per year through shark and ray-related tourism. In over 20 years, the Bahamas have offered over 1 million shark-diver interactions, contributing an estimated gross of US $800 million to the Bahamian economy (Cline 2008).

Threats (see Appendix for additional information)

The Caribbean Reef Shark is targeted and/or captured as bycatch in artisanal and commercial gillnets and longlines throughout its range. Artisanal fisheries are intense across much of Atlantic South America, and there are largely unmanaged commercial longline fisheries in several areas. In Caribbean Colombia, artisanal fisheries are widespread and lack management, and this shark is frequently captured in the demersal longline fishery, particularly off San Andrés Archipelago (Gomez *et al.* 2017). In

Venezuela, commercial and artisanal fisheries are intense, they lack management, and have exhibited peaks in catches followed by declines, indicative of sequential overfishing (Mendoza 2015). This shark is captured in large numbers there off the Los Roques Archipelago, including juveniles and neonates (Tavares 2009). Artisanal fisheries are likely to capture this species across the Guianas, but there are no data. In northwestern Brazil, artisanal fisheries pressure is high and 44% of target stocks were likely to be overfished by the end of the 2000s (Vasconcellos *et al.* 2011). The combination of intense and unmanaged artisanal and commercial fishing in that area has led to the disappearance of several elasmobranch species in the region, including Smalltooth Sawfish (*Pristis pectinata*), Largetooth Sawfish (*Pristis pristis*), Daggernose Shark (*Isogomphodon oxyrhynchus*), and the congeneric Smalltail Shark (*Carcharhinus porosus*) (Charvet and Faria 2014, Lessa *et al.* 2016, Reis-Filho *et al.* 2016, Santana *et al*. 2020). In northeastern and eastern Brazil, artisanal fisheries are intense, gillnetting is the predominant artisanal gear, fishers there report that stocks are over-exploited, and other sharks have been depleted (Guebert-Bartholo *et al.* 2011, Reis-Filho *et al.* 2014). In southern Brazil, artisanal fisheries are intense, and 58% of stocks targeted by artisanal fishers are overexploited, half of those having collapsed (Vasconcellos *et al.* 2011). In addition to fisheries pressure, coral reef ecosystems in the Caribbean Sea, which are the primary habitat for this species, are in decline due to climate change, specifically coral bleaching, disease, invasive species, and coastal pollution (Carpenter *et al.* 2008, Jackson *et al.* 2014). Overall, this shark has very little refuge from fishing, and is at threat from a continuing decline in habitat quality.

Conservation Actions **(see Appendix for additional information)**

In July 2006, the United States National Marine Fisheries Service finalized the Consolidated Atlantic Highly Migratory Species Federal Management Plan. This Federal Management Plan includes a range of management measures including quotas and seasonal closures. Caribbean Reef Sharks are listed as a prohibited species to commercial and recreational harvest under this management plan.

In the Bahamas, longline fishing was banned in 1993, and peak elasmobranch export occurred the same year. The export steadily declined until the last export of any elasmobranch product from The Bahamas occurred in 2004. The Bahamas Shark Sanctuary was established in 2011 (Haas *et al*. 2017), and it is currently illegal to fish for, land or possess any sharks or shark parts there. Enforcement is very underfunded and inadequate, making it possible that exploitation may be occurring at present or in the future.

Cuba has designated 21% of the country as a Marine Protected Area. There is no fishing for sharks or rays in these areas. The most important Marine Protected Area is Jardines de la Reina, Alejandro de Humbolt, Guanacabibes. Shark fishing is illegal in Jardines de la Reina Marine Reserve and local guides have noted an increase in sightings of many species of reef-associated sharks including this species since 1997 when the reserve was declared and enforcement of fishing regulations began. Cuba also has two places designated for diving with sharks, Santa Lucia and Jardines de la Reina. Cuba has a prohibition on finning and in 2015 adopted a National Plan of Action for Sharks (NPOA – Sharks).

In Mexico, there is a closed season for the shark fishery in May and June in the states of Tamaulipas, Veracruz and Quintana Roo, and May 15–June 15, and August 1–29 in the states of Tabasco, Campeche and Yucatán. The Official Standard NOM-029-PESC-2007 establishes management measures that finning be prohibited. It also establishes that fishing is prohibited at the surrounding area of 5 km of coral reef areas, at adjacent areas in front of mouth rivers and coastal lagoons (delimited by a semicircle of 2.5 km

in the mouth), and at nursery areas (prohibited the use of gill-nets in June) in front of Playa Bagdad, Tamaulipas. There is a marine protected area of 30 km from the Rio Bravo´s mouth to Conchinal; Terminos lagoon, Campeche; Usumacinta-Grijalva Rivers, Tabasco; Yalahau lagoon, Quintana Roo; and Espiritu Santo and Ascension Bays, Quintana Roo. There is also a Management Plan of the Natural Protected Area "Biosphere Reserve of the Mexican Caribbean" that prohibits any elasmobranch fishery in all this biosphere reserve that comprises all Mexican Caribbean. There are enforcement problems.
In the wider Caribbean, the Regional Office of Fisheries of the System of Integration (SICA), prohibits shark finning with the resolution OSP-05-11 in all of the Central American countries from Belize to Panama, including the Dominican Republic.

In Belize, there are currently no conservation measures protecting sharks throughout territorial waters. Despite no-take restrictions in marine reserves, illegal fishing continues to take place and reef sharks are known to be caught. As such, increased enforcement is required in Belize's 13 marine reserves to specifically protect shark species against illegal fishing. Most sharks caught as bycatch in the hook and line fisheries could be released live but are usually landed.

In Guatemala, there is a fishing ban for one month (usually August) for the capture of sharks (Families: Carcharhinidae, Sphyrnidae, Triakidae, and Ginglymostomatidae) and two months (usually July and August) for rays (Family Dasyatidae). However, enforcement of these bans is a challenge.

In Honduras, there is a national regulation (Governmental Decree 107-2011) that declared the territorial waters of Honduras a shark sanctuary. Fishing and exploitation of all shark species was prohibited in all territorial waters of Honduras. This regulation was revised by a new regulation (Governmental Decree 26-2016), which indicated that sharks that are incidentally caught can be used. There are enforcement problems.

Currently, there are some marine protected areas in the middle and west of the Caribbean coast of Panama (e.g. Portobelo National Park, Bastimentos Island National Marine Park). These protected areas provide a certain level of ecosystem management through implementing control over local fisheries and providing protection for critical habitats (e.g. coral reefs, mangrove), both of these measures can benefit shark populations. Despite the creation of these marine parks, enforcement of protective initiatives in these areas is a challenge. In addition to the creation of marine protected areas, Panama has established nationwide conservation and management policies for the fisheries industry. In 2006, an official regulation was approved that prohibits shark finning in Panama waters. In 2017, new regulations were implemented for longline fisheries (e.g. limits of the number of hooks to 1,000 per surface longline strand), which will help to further contribute to the conservation and management of sharks in Panama.

The archipelago of San Andrés y Providencia, Colombia, was declared a Biosphere Reserve by UNESCO in 2000, and covers an area of 180,000 km^2 (i.e., 10% of the total area of the Caribbean Sea) and comprises some of the most productive and biodiverse reefs in the region. Since 2008, all shark species have been fully protected in the Seaflower Biosphere Reserve, banning fishing and possession of all sharks and their byproducts. These has led to one of the highest shark abundances found in the Caribbean Sea. Local communities pushed for these shark conservation measures in the Archipelago, simplifying the enforcement by local authorities. Enforcement in remote regions against poaching has been challenging, although the Colombian Navy has been leading the enforcement of these areas and has enabled scientific data collection. In addition, Colombia has banned wire-leaders in their longlining fisheries as a bycatch-reduction measure across the entire Exclusive Economic Zone (EEZ), where industrial fisheries are only allowed to have a total shark by catch of 35% in a single fishing trip. Finning is prohibited within the Colombia EEZ and fins are required to be attached naturally to the body at the point of landing.

Antilles, Antigua and Barbuda, Barbados and Trinidad and Tobago developed draft NPOA – Sharks in 2015, 2016 and 2017, respectively. However, for Barbados and Trinidad and Tobago, these are still in draft form due to a combination of factors including lack of political will. For the Dutch Caribbean, spatial management measures have been implemented. The Yarari Marine Mammal and Shark Sanctuary was created in 2015 for Bonaire, St. Eustatius and Saba in the Caribbean Netherlands. In November 2011, the Kingdom of the Netherlands ratified the Memorandum of Understanding on the Conservation of Migratory Sharks (MoU Sharks) of the Convention on the Conservation of Migratory Species. There are no species-specific protections or conservation measures in the South American portion of its range. In Brazil, there has been a finning ban in place since 1998 and a National Plan of Action (NPOA) for sharks and rays was implemented in 2014.

To conserve the population and to permit recovery, a suite of measures will be required which may include species protection, spatial management, bycatch mitigation, and harvest and trade management measures (including international trade measures). Effective enforcement of measures will require ongoing training and capacity-building (including in the area of species identification). Catch monitoring is needed to help understand population trends and inform management.

Credits

Assessor(s):	Carlson, J., Charvet, P., Blanco-Parra, MP, Briones Bell-lloch, A., Cardenosa, D., Derrick, D., Espinoza, E., Morales-Saldaña, J.M., Naranjo-Elizondo, B., Pérez Jiménez, J.C., Schneider, E.V.C., Simpson, N.J., Talwar, B.S., Crysler, Z., Pacoureau, N. & Kyne, P.M.
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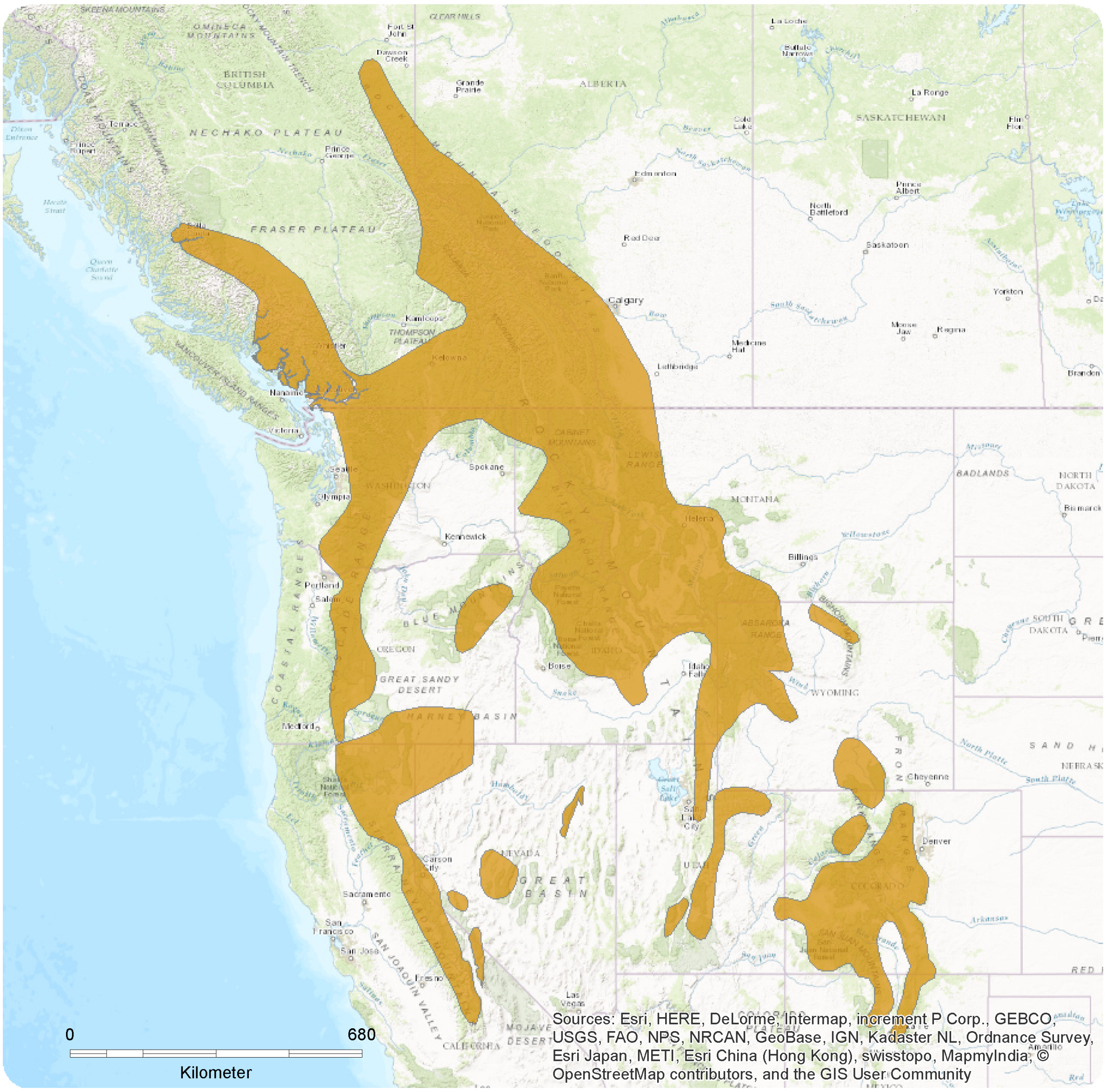
Authority/Authorities:	IUCN SSC Shark Specialist Group (sharks and rays)

Distribution Map - American Pika

Ochotona princeps



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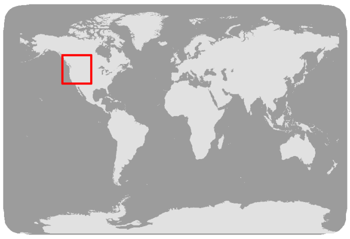
Range

🟧 Extant (resident)

Compiled by:

IUCN (International Union for Conservation of Nature)

NE DD **LC** LEAST CONCERN NT VU EN CR EW EX



Population

American Pikas are individually territorial (male and female territories are of approximately similar sizes; Smith and Ivins 1984) on a very discrete and identifiable habitat type, talus or piles of broken rock. Although territory size may vary seasonally and by habitat quality, a rough average from several detailed studies (reviewed in Smith and Weston 1990) is approximately 500 m^2. This size also roughly corresponds to the area as determined by the average nearest-neighbour distance (diameter of a pika territory; area determined by πr^2) between centres of activity of individuals (basically, the average distance between haypiles; Smith and Weston 1990). Thus in prime talus habitat, about 20 pikas may occur per hectare.

As highlighted in the discussion of Distribution (above), pika populations are declining in some parts of the species' geographic range, primarily at low elevations in the Great Basin (populations being lost, and other populations shifting their distribution upslope). From everything we know about pika biology (thermally sensitive) and dispersal ability (extremely limited, especially in warm environments; MacArthur and Wang 1973, 1974; Smith 1974a,b), it is unlikely that any of these lost populations can be resettled within ecological time – thus they represent a loss and we conclude that the global population is decreasing.
Current Population Trend: Decreasing

Habitat and Ecology (see Appendix for additional information)

The American Pika is a candidate indicator species for the effects of climate change (in particular, global warming, although interactions with precipitation and snowpack merit continuing attention), because it occurs in a very specific insular habitat type, has a very low reproductive rate (counterbalanced by being relatively long-lived for a small mammal), exhibits very limited dispersal ability, and is primarily diurnal and extremely sensitive to warm temperatures.

The American Pika is a typical rock-dwelling species of pika (Smith *et al.* 1990). It primarily inhabits talus and talus-like formations adjoining a meadow or source of vegetation in cool and moist microclimates across western North America (Smith and Weston 1990, Hafner 1994, Millar and Westfall 2010). Talus habitat is typically insular or patchy in nature at several spatial resolutions (Smith 1974a, Smith and Gilpin 1997). Pikas prefer talus in RIF (rock-ice-feature) formations (83% of sites in one study of 420 pika sites), and with rock diameters of 0.2-1.0 m (Tyser 1980, Hafner 1994, Beever *et al.* 2010, Millar and Westfall 2010). They may also occur in lava flows and anthropogenic habitats such as mine ore dumps or road cuts; occasionally they may live in piles of logs or similar habitat (Smith 1974a,b; Millar and Westfall 2010; Rodhouse *et al.* 2010; Manning and Hagar 2011).

American Pikas are individually territorial on talus habitat (as calculated above; at a density approximating 20 individuals per hectare). Pikas are relatively long-lived for a small mammal (120–175 g); some pikas may live to the age of 6 or 7 years, and many live to the age of 3-4 years (Millar and Zwickel 1972a; Smith 1978). The reproductive rate of American Pikas is low; females initiate two litters per year, although it is most common for only one litter to be weaned successfully (Smith 1978; Smith and Ivins 1983a; Smith and Weston 1990). Average litter size at time of parturition ranges from 2.3–3.7 (range 1–5 young), although there is an erosion of fecundity during weaning such that most females only successfully wean two (or at the most three) young per year (Smith 1978; Smith and Weston 1990).

Mortality rate is highest in the juvenile age class (Millar and Zwickel 1972a; Smith 1978). Juveniles must claim a vacant territory to survive the winter, and in saturated populations availability of vacant territories is contingent on the low mortality rate of adults (Smith 1978; Smith and Ivins 1983b).

In ecological studies where pikas have been individually marked, American Pikas have consistently been found to be poor dispersers (Tapper 1973, Smith and Ivins 1983b, Smith 1987, Peacock 1997). It is rare for an adult to disperse; once settled on a territory, they tend to remain there for life (the occasional dispersal movement by an adult is to move to a vacant site adjoining its home territory if it is of substantially higher quality; Smith and Ivins 1983b; Smith 1987). Juveniles tend to remain philopatric, largely occupying space in the interstices between adult territories. Juveniles also time-share activity periods with resident adults; they are primarily active when adults are inactive. This strategy ensures that juveniles are familiar with a region of talus so that they can claim any territory vacancy that may appear (Smith and Ivins 1983, 1987; Tapper 1973; Smith 1987). As a result, few juveniles attempt to disperse away from their birthplace. When juveniles attempt to disperse within a talus patch, they are socially rebuked by non-parental adults; when they leave a patch in an attempt to find available talus elsewhere, they increase their exposure to predators and are unable to use the safety of the talus to cool themselves if the ambient conditions are too warm (Smith 1974b, Smith and Ivins 1983, Smith 1987). Ability of juveniles to engage in long-distance (intra-patch) dispersal appears related to temperature. At low elevations dispersal distance and probability of success are low (Smith 1974 a,b; Peacock and Smith 1997; Smith and Gilpin 1997). Most observed long-distance dispersal has occurred at higher elevations where the talus tends to be more continuous and it is cooler (Tapper 1973; Smith 1974 a,b; Smith 1987; Peacock 1997). An understanding of pika dispersal is necessary because most observed pika population extirpations have occurred at low (hot) elevations, and it is under these conditions that probability of subsequent re-colonization of these sites is extremely low. When a high-elevation population temporarily disappears, its chance of being recolonized is significantly greater (Tapper 1973, Smith 1987).

Pikas are very sensitive to warm or hot temperatures; when daytime temperatures are hot during summer, they tend to avoid the warmer temperatures and concentrate their activity at dawn and dusk (MacArthur and Wang 1974, Smith 1974b). Under these conditions, they may even exhibit nocturnal behaviour (Smith 1974b). Pikas have a relatively high body temperature and a relatively low upper lethal temperature; thus, they have very little flexibility with regard to physiological temperature regulation. Field experiments have shown that when confined in the sun and unable to behaviourally thermoregulate (such as by darting into the interstices of the talus where it is always significantly cooler), they can die at relatively low ambient temperatures (25.5 – 29.4oC)(MacArthur and Wang 1973, 1974; Smith 1974b). This temperature sensitivity puts dispersing pikas, particularly at lower, warmer elevations, at increased risk and is why restricted dispersal distances under these conditions are most commonly observed (Smith 1974a,b). Relative vulnerability of pika populations to climatic stress may be indexed by variables such as latitude and elevation, two factors known to affect local climate and, in turn, the distribution of American Pikas (Grinnell 1917, Smith 1974a, Beever *et al.* 2011).

Pikas are generalized herbivores. As the American Pika does not hibernate, it must collect food during summer that it stores in a haypile or cache in the talus to serve as food during winter. Thus, during the summer the pika has two distinctly different foraging strategies: the direct consumption of food and haying behaviour (Huntly *et al.* 1986). Haying reaches a crescendo in mid-late summer; during this time a pika may make hundreds of trips each day off the talus to clip and harvest plants for its haypile (Smith and Ivins 1984). The plants available to a pika to eat or hay are restricted to the vegetation on or adjoining its territory. Pikas are highly selective of which plants they consume or harvest, choosing plants that are higher in water content, protein, and select micronutrients (West 1981, Millar and Zwickel 1972b) and other characteristics (reviewed in Smith and Weston 1990). The timing of haying is related to the phenology of plants at any given altitude (that is, they begin and end this activity earlier at lower elevations), gender (males generally initiate haying earlier) and age (adults begin haying before juveniles) (Smith 1974b, Smith and Weston 1990).

Systems: Terrestrial

Use and Trade

The American Pika is not utilized or involved in trade of any sort. Its primary economic importance is as a focus for ecotourism; thus, it may support livelihoods of local people.

Threats (see Appendix for additional information)

The most pervasive threat affecting the American Pika appears to be contemporary climate change. While the species as a whole is Least Concern, large tracts of its extent of occurrence (EOO)—namely across the Great Basin – have seen local population extirpations, range contractions (upslope movement of the lower-elevation range boundary), and reduced densities (Beever *et al.* 2003, 2010, 2011; Wilkening *et al.* 2011). Three alternative classes of direct thermal stresses may affect the persistence of pika populations (Beever *et al.* 2010): 1) acute cold stress (number of days below a very cold threshold temperature); 2) acute heat stress (number of days above a warm threshold temperature; and 3) chronic heat stress (such as average summer temperature). Acute cold stress may arise from reduced snowpack (observed across western North America; Mote *et al.* 2005) which in turn may result from more precipitation falling as rain and less as snow, or warmer days leading to a more rapid melt-off of snowpack, or both. Without the snowpack that acts as an insulator during winter, pikas might freeze or extinguish their food supplies while attempting to thermoregulate leading to an increase in mortality (Smith 1978, Beever *et al.* 2010). While some areas have extant populations at higher elevations to and from which pikas could disperse, the insular nature of talus habitat means that many populations that become extirpated will remain so on an ecological time scale – the likelihood of recolonization is extremely remote. While extinctions in the Great Basin have been going on for a long time (~7,500 years), the recent rate is high and indicative of observed increases in temperature in the region due to climate change (Beever *et al.* 2011). Current American Pika distributions in the Great Basin represent a perfect storm generated by the high temperatures (in summer) and reduced snowpack (in winter; Mote *et al.* 2005) caused by climate change.

A counterbalance to the data accumulated in the Great Basin is the population of pikas at Bodie, California, at about 2,550 m elevation just 35 km west of the Sierra Nevada crest where summer temperatures are relatively warm. There pikas occupy insular patches of habitat (ore dumps left by mining activity) that are spaced across a landscape of Great Basin shrub vegetation. This classic metapopulation, the longest-term study of any pika species, has been observed since the late 1940s (Severaid 1955) and semi-continuously since 1969 (Smith 1974 a, b, 1978, 1980; Smith and Gilpin 1997), including annual censuses (with a couple of two-year gaps) from 1989 to 2010 (Smith unpublished data). Approximately 78 ore dump patches have been included in each census. In 1991 the southern half of the figure-eight-shaped study area experienced a meta-population collapse and has not recovered

(Smith and Gilpin 1987); in 2010 the southern constellation of patches was still void of pikas. However, the 37 patches in the northern constellation showed a slightly higher occupancy rate in 2009 (84%; the lowest percent occupancy was 49% in the north) than in the first full census in 1972. Thus, there appears to be no evidence that heat stress in summer at Bodie causes mortality or population decline of pikas on these small habitat islands, although warm temperatures may have inhibited colonization of the southern constellation.

Another largely Great Basin threat is competition with free-ranging and feral cattle in those situations where livestock are allowed to graze within the typical pika foraging distance from the talus margin (Beever *et al.* 2003). Most pikas live in cattle-free areas or, in Colorado, grazing temporally occurs primarily in fall after pikas have completed their haypiles. But in the Great Basin cattle graze adjoining many of the limited pika populations. Beever *et al.* (2003) noted that cattle grazing could be contributory to some pika population extirpations. Pika density was lower in areas that were heavily grazed—primarily small sites with more edge compounding the effect of grazing (Beever unpublished data). Further, presence of livestock grazing increased in importance from the 20th century observations to the 1999-2008 observation period in terms of predicting the pattern of site-level persistence of pikas in the Great Basin (Beever *et al.* 2011). Millar (in press) has compared placement of pika haypiles in Great Basin ranges at sites with and without cattle grazing. Normally, pikas place their haypiles close to the talus/vegetation interface; at her cattle-free sites this distance averaged 1.8 m from the talus edge. At grazed sites pikas placed their haypiles an average of 30 m upslope from the talus edge and were forced to forage on comparatively poor vegetation growing amidst the rocks. Millar concluded that grazing effects could be contributing to observed regional differences in viability of pikas.

Conservation Actions **(see Appendix for additional information)**

• Pikas occur in many national parks and other protected areas throughout their range in the United States and Canada. Pikas tend to occupy areas away from human habitations or influence, and they appear to not be negatively influenced by trails or nearby roads that do encroach on their habitat (for example, they often colonize road cuts (Manning and Hagar 2011, Millar and Smith pers. obs). Neither hunting nor trapping of American Pikas is allowed throughout their range.
• Additional research on the potential competitive relationship between livestock grazing and pikas in the Great Basin should be initiated, and if it is shown that competition between pikas and livestock occurs and increases the probability of local extirpation of Great Basin pika populations, management plans should be enacted to eliminate grazing in areas adjoining known pika populations.
• The American Pika should be considered an early-warning indicator species for the effects of climate change and continuously monitored throughout their range to demonstrate how a species with these attributes could adapt and be resilient in the face of climate change, or to determine their decline and how it could be reversed. Particular attention should be given to the establishment of scientifically valid protocols for 1) long-term monitoring of populations; 2) parsing out the relative contributions of acute cold stress, acute heat stress and chronic heat stress on pikas given climate change throughout the range of the species; 3) monitoring of pika behaviour with relation to micro-climates present in their environment; 4) determining how food selectivity indices may vary across the range of the species and how these may be affected by climate change; 5) determining any measurable manifestations of physiological stresses on pikas, such as disease, increased levels of stress hormones, reduced reproductive capacity, reduced longevity, etc., as a result of climate change; 6) understanding the relationships of American Pikas with other syntopic and sympatric species such as marmots (*Marmota*

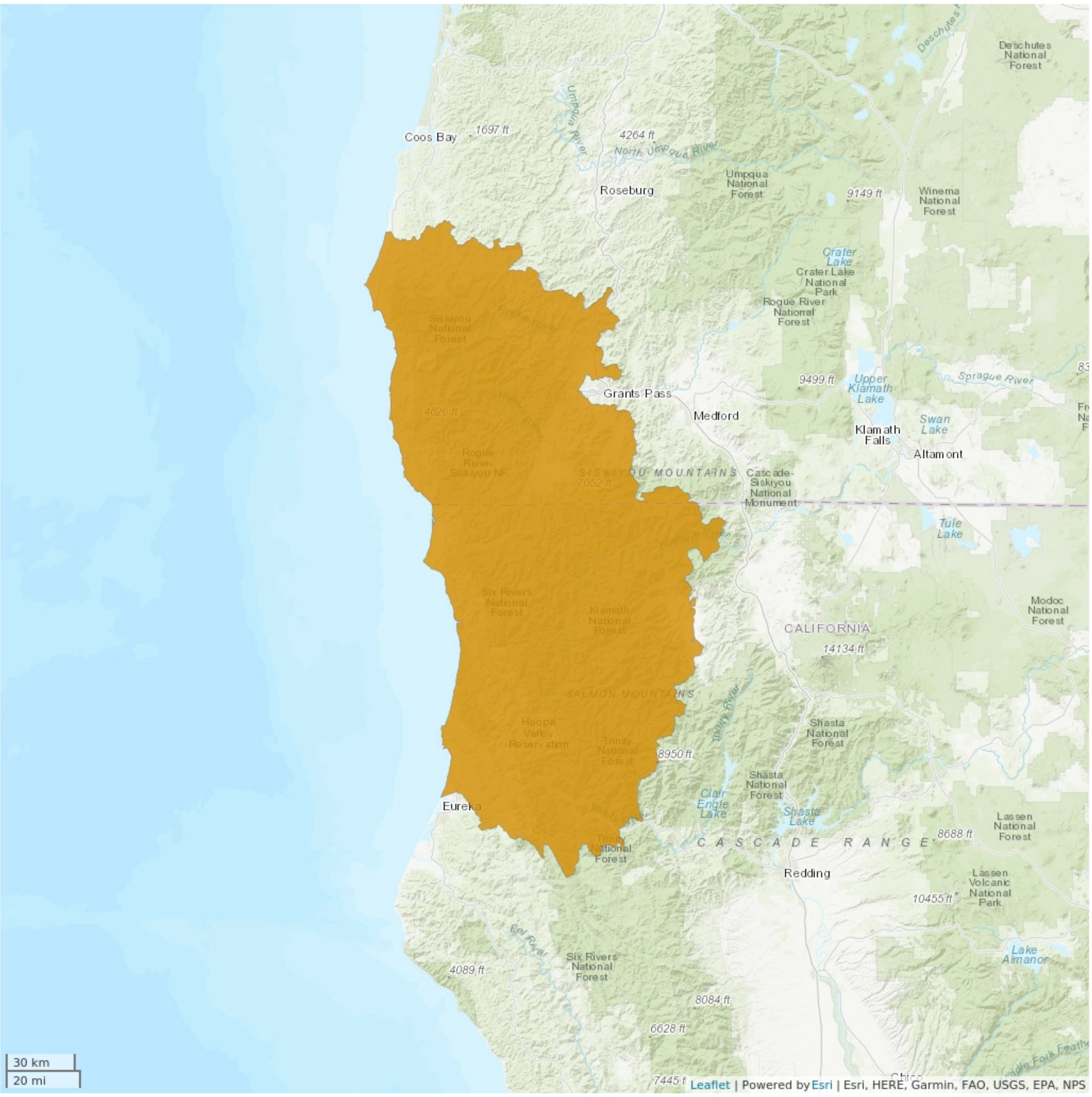
spp.), woodrats (*Neotoma* spp.), chipmunks (*Tamias* spp.), etc.; and 7) understanding the additive or synergistic roles that multiple types of climate stress can have in concert (such as lower growing season precipitation combined with warmer summer average temperatures).

- Particular attention should be given to examine and inventory sites of accurate historical low-elevation records of occurrence of pikas, as contemporary observations at these sites can give us a quantitative measure of potential change over time in the distribution and abundance of American Pikas with regard to climate change.
- If it is shown that climate change is negatively influencing the American Pika range-wide and potentially endangering the species, accommodation, mitigation and active conservation strategies should be enacted at the regional, national and international scales.

Credits

Assessor(s):	Smith, A.T. and Beever, E.
Reviewer(s):	Battistoni, A.

Distribution Map - Del Norte Salamander



Legend

🟧 EXTANT (RESIDENT)

Compiled by:

IUCN (International Union for Conservation of Nature), Conservation International & NatureServe. 2020

Population

This species is locally abundant in appropriate habitat. However, it is thought to be declining across portions of its range that are being logged due to the absence of federal protection (Welsh and Bury 2005).

Current Population Trend: Decreasing

Habitat and Ecology (see Appendix for additional information)

This species is strongly associated with moist talus and rocky substrates in redwood or Douglas fir forests, including in riparian zones. It is usually found among moss-covered rocks, under associated bark and other forest litter, but usually not in seeps or other very wet areas. It is associated with older forests with closed, multi-storied canopy (composed of both conifers and hardwoods), with a cool, moist microclimate, and rocky substrates dominated by cobble-sized fragments, and these conditions may be optimal throughout most of the range (Welsh 1990, Welsh and Lind 1995). In damp coastal regions, the presence of enclosed canopy-forest may be less of a requirement as this species may be common in recently harvested forest areas with no associated older forests (Diller and Wallace 1994, Green *et al.* 2014). The eggs are laid in concealed terrestrial sites where they develop directly without a larval stage.

Systems: Terrestrial

Use and Trade (see Appendix for additional information)

There are no records of this species being utilized.

Threats (see Appendix for additional information)

This species is thought to primarily be threatened by clear-cut logging of old-growth and mature forest habitat, especially in drier inland locations (Welsh and Lind 1988, 1992, Welsh and Bury 2005). In wetter areas, this species does not appear to be negatively affected by logging (Diller and Wallace 1992). This threat is particularly pervasive on private timberlands, highlighting the importance of National Forest Service land (Moeur *et al.* 2011, Davis *et al*. 2013, Evelyn and Sweet 2018). Additional threats include wildfires, climate change, and the potential future threat of disease. Large wildfires, combined with the anticipated higher temperatures and change in hydroperiod from climate change, could further alter and reduce the availability of moist microhabitats utilized by this species. Additionally, the increase in frequency of large fires poses a threat, as wildfires pose a substantial threat to the habitat of this species by reducing overstory cover and degrading forest floor conditions through loss of suitable cover, loss of invertebrate prey species, and direct desiccation of the spoil (Evelyn and Sweet 2018). Regarding disease, while *Bd* has not been found in this species, it is known to be present in Oregon and Plethodontid salamanders in holarctic climates are known to carry the disease (Weinstein 2009, Evelyn and Sweet 2018).

The emergence of the salamander chytrid fungus (*Batrachochytrium salamandrivorans*, "*Bsal*") as an amphibian pathogen has caused dramatic declines in European fire salamanders since its apparent arrival in 2008 through the international pet trade. Although the presence of the fungus has not been confirmed in the Americas, it is spreading in Europe and risks spreading further. Within the US, the Pacific coast has been identified as a potentially high risk area for *Bsal* introduction and spread (Yap *et* *al.* 2015, Richgels *et al.* 2016). Members of the families *Plethodontidae* and *Salamandridae* are particularly susceptible to *Bsal* (Martel *et al*. 2014).

Conservation Actions (see Appendix for additional information)

Conservation Actions In-Place

This species occurs in Redwood National Park and the associated state redwood parks in Humboldt and Del Norte Counties, California. It is also found in the Klamath, Shasta Trinity and Six Rivers National Forests in California, Rogue River-Siskiyou National Forest in Oregon, BLM Arcata Field Office in California, and BLM Coos Bay, Roseburg, and Medford Districts in Oregon. Overall, 87% of recorded sites occur on Forest Service, Bureau of Land Management and National Park Service lands, with 13% of sites occurring on other lands. The sites can be further divided into land allocation types that signify three levels of land use intensity: resource base, moderate disturbance, and minimal disturbance. 13% of sites are considered unknown, as they do not occur on federal lands. 44% of sites occur on sites with minimal disturbance, 7% experience moderate disturbance, and 36% are in resource base land use allocations (C. Hughes pers. comm. February 2021). This species was previously afforded protection from ground-disturbing activities on federal lands under the Northwest Forest Plan, but is no longer listed as a Survey and Manage Species under this plan (Green *et al.* 2014). While the species no longer receives specific protections under the Survey and Manage standards and guidelines, the reserve network of the North West Forest Plan and other standards and guidelines were deemed to afford the species a reasonable assurance of persistence without listing, which is what prompted the species' removal from Survey and Manage status in 2001 (C. Hughes pers. comm. February 2021).

In an effort to prevent the introduction of *Bsal* into the US, an Interim Rule of the Lacey Act has been enacted that bans the importation of 201 species of salamanders (USFWS 2016). Additionally, a temporary voluntary trade moratorium of imports of Asian salamander species that are known to carry the disease until such time as effective testing and treatment regimens can be developed and distributed has been recommended to all exporters, shippers, sellers and buyers by the Pet Industry Joint Advisory Council (PIJAC). A North America *Bsal* Task Force has also been created, with working groups designed to address a variety of disease prevention and mitigation goals (North America *Bsal* Task Force 2021).

Conservation Needed

It is recommended that this species be reconsidered for protection as a Survey and Manage Species, as the population is thought to have declined as a result of timber harvest activities (Welsh and Bury 2005). Additionally, continued and strengthened management of protected area(s) where this species occurs, and expanded protection of suitable habitat elsewhere in its range is also recommended.

Research Needed

More information is needed on this species' distribution, population status, ecology, and threats.

Credits

Assessor(s): IUCN SSC Amphibian Specialist Group

Reviewer(s): Bowles, P.

Distribution Map - Grey Wolf

Leaflet | Powered by Esri | RJGC, Esri, HERE, FAO, NOAA, AAFC, NRCan

Legend

- 🟧 EXTANT (RESIDENT)
- 🟩 EXTANT & REINTRODUCED (RESIDENT)
- 🟥 EXTINCT

Compiled by:

IUCN (International Union for Conservation of Nature) 2023

Population

Because of the diversity in climate, topography, vegetation, human settlement and development of wolf range, wolf populations in various parts of the original range vary from extinct to relatively pristine. Wolf densities vary from about one/12 km² to one/120 km².

Mech and Boitani (2004) provide details, for each range country, on subspecies present, population status, approximate numbers, the percentage of former range occupied at present, main prey (where known), legal status, and cause of decline. For Europe, the wolf population is essentially a large meta-population with several distinct subpopulations; the status and trends for each population is provided by the Large Carnivore Initiative for Europe (http://www.lcie.org/) and in Chapron *et al*. (2014). The status and trends for wolf populations in the United States are provided by the U.S. Fish and Wildlife Service and game agencies in several US states, including Minnesota, Wisconsin, Michigan, Montana, Idaho, Wyoming, Oregon, Washington, Arizona, and New Mexico. In general, population increases in North America and Europe are likely to be off-set by localized declines in other parts of the range.
Current Population Trend: Stable

Habitat and Ecology **(see Appendix for additional information)**

Ranges in all northern habitats where there is suitable food (Mech 1970), densities being highest where prey biomass is highest (Fuller 1989). Food is extremely variable, but the majority comprises large ungulates (moose, caribou, deer, elk, wild boar, etc.). Wolves will also eat smaller prey items, livestock, carrion, and garbage.

Systems: Terrestrial

Use and Trade

There is sustainable utilization of the species' fur in Canada, Alaska, Montana and the former Soviet Union and Mongolia, although illegal hunting for fur does occur in some range states.

Threats **(see Appendix for additional information)**

The historical range of the Grey Wolf has been reduced by about one-third, primarily in developed areas of Europe, Asia, Mexico, and the United States by poisoning and deliberate persecution due to depredation on livestock. Since about 1970, legal protection, land-use changes, and rural human population shifts to cities have arrested wolf population declines and fostered natural recolonization in parts of Western Europe and the United States, and reintroduction in the western United States and Mexico. Continued threats include competition with humans for livestock, especially in developing countries, exaggerated concern by the public concerning the threat and danger of wolves. Fragmentation of habitat, with resulting areas becoming too small for populations with long-term viability can be a threat in parts of the range with highest density of human activities.

Conservation Actions **(see Appendix for additional information)**

Legislation

The species is included in CITES Appendix II, except populations from Bhutan, India, Nepal and Pakistan, which are listed on Appendix I. The species is strictly protected under the Bern Convention (Appendix II) and Habitats Directive (Annex II and IV), and there is extensive legal protection in many European countries; however, there are national exceptions, and enforcement is variable and often non-existent.

Presence in protected areas

The Grey Wolf occurs in many protected areas across its range.

Occurrence in captivity

The species lives and breeds well in captivity and is common in many zoological gardens.

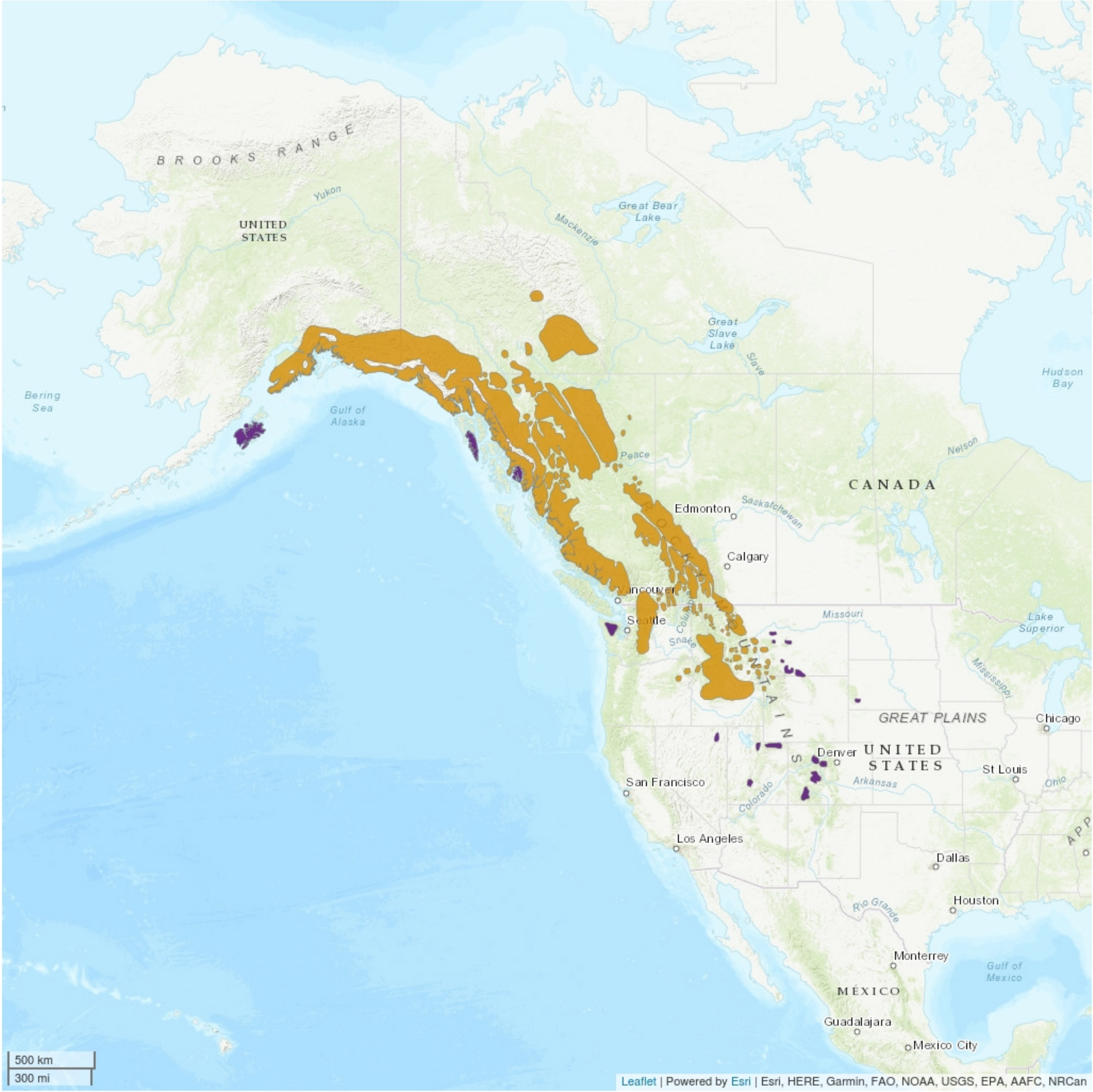
Gaps in knowledge

One of the most important questions still remaining about wolves involves the nature of their interaction with prey populations. The conditions under which wolves limit, regulate, or control their population is still open and important (Mech and Boitani 2003). Of recent and increasing importance is clarifying the role that wolves play in precipitating ecosystem effects, particularly trophic cascades (Mech and Boitani 2003, Ripple and Betscha 2004, Peterson *et al*. 2014). Of more academic interest are questions involving wolf genetics, scent-marking behaviour, pseudo pregnancy and diseases (Mech 1995). Of recent and increasing importance is the issue of hybridization with dogs (and indeed other *Canis* spp) in several parts of the range in Europe and Asia, with unknown impact on medium- and long-term genetic integrity of the species.

Credits

Assessor(s):	Boitani, L., Phillips, M. & Jhala, Y.
Reviewer(s):	Hoffmann, M. & Sillero-Zubiri, C.
Contributor(s):	Mech, D.
Facilitator(s) and Compiler(s):	Hoffmann, M.
Authority/Authorities:	IUCN SSC Canid Specialist Group (foxes, jackals and wild dogs)

Distribution Map - Mountain Goat



Legend

🟧 EXTANT (RESIDENT)

🟪 EXTANT & INTRODUCED (RESIDENT)

Compiled by:

IUCN (International Union for Conservation of Nature) 2024

Population

In 2010, the total population in Canada was estimated at 43,700 to 70,200, distributed as follows: Alberta 2000; British Columbia 39,000-65,500; Northwest Territories 1,000 (earlier estimate); and Yukon 1,700. Recent total estimates in the United States are 37,000 to 47,000 individuals, with more than 13,000 animals in the contiguous states, and 24,000 to 33,500 in Alaska. (Myatt and Larkins 2010)

Some historic ranges in southern part of distribution are now unoccupied. These historic declines were likely due to overharvest, partly because managers initially did not realize that this species can only tolerate a very small amount of harvest in native populations.
Current Population Trend: Stable

Habitat and Ecology (see Appendix for additional information)

Most mountain goats occur in high altitude habitats, up to the limit of vegetation. Although they descend to sea level in coastal areas, they are primarily an alpine and sub-alpine species. Throughout the year, the animals usually stay above timberline, but they will migrate seasonally to higher or lower elevations within that range. Summertime movements to low-elevation mineral licks often take them several or more kilometers through forested areas. Their diet includes grasses, forbs, sedges, ferns, moss, lichen, twigs, and leaves from the low-growing shrubs and conifers of their high-altitude habitat. Their main predators are cougars, wolves and brown bears. The species lives in groups that can vary from a few to a hundred, and during the winter months groups generally coalesce to form large herds. The gestation length is approximately 180 days, with a single birth typically, although twins are common in introduced populations. The average age of primiparity varies from 2 to 5 years among populations. Male mortality is higher than female mortality and very few males live longer than 10 years. Very few females survive more than 16 years (Festa-Bianchet and Côté 2008).

Systems: Terrestrial

Use and Trade

Hunting is now well managed in most jurisdictions. Changes in management regimes have stabilized past declines.

Threats (see Appendix for additional information)

These animals are largely protected from threats due to the inaccessible nature of their habitat. The species is hunted, but regulations in both of its range states have stabilized past declines. Mountain goats are more sensitive to human disturbance than most other ungulates, and are particularly sensitive to harassment from aircraft. Increasing aircraft use, particularly helicopters, for industrial and recreational purposes on mountain goat habitat is a major concern for their conservation.

Conservation Actions (see Appendix for additional information)

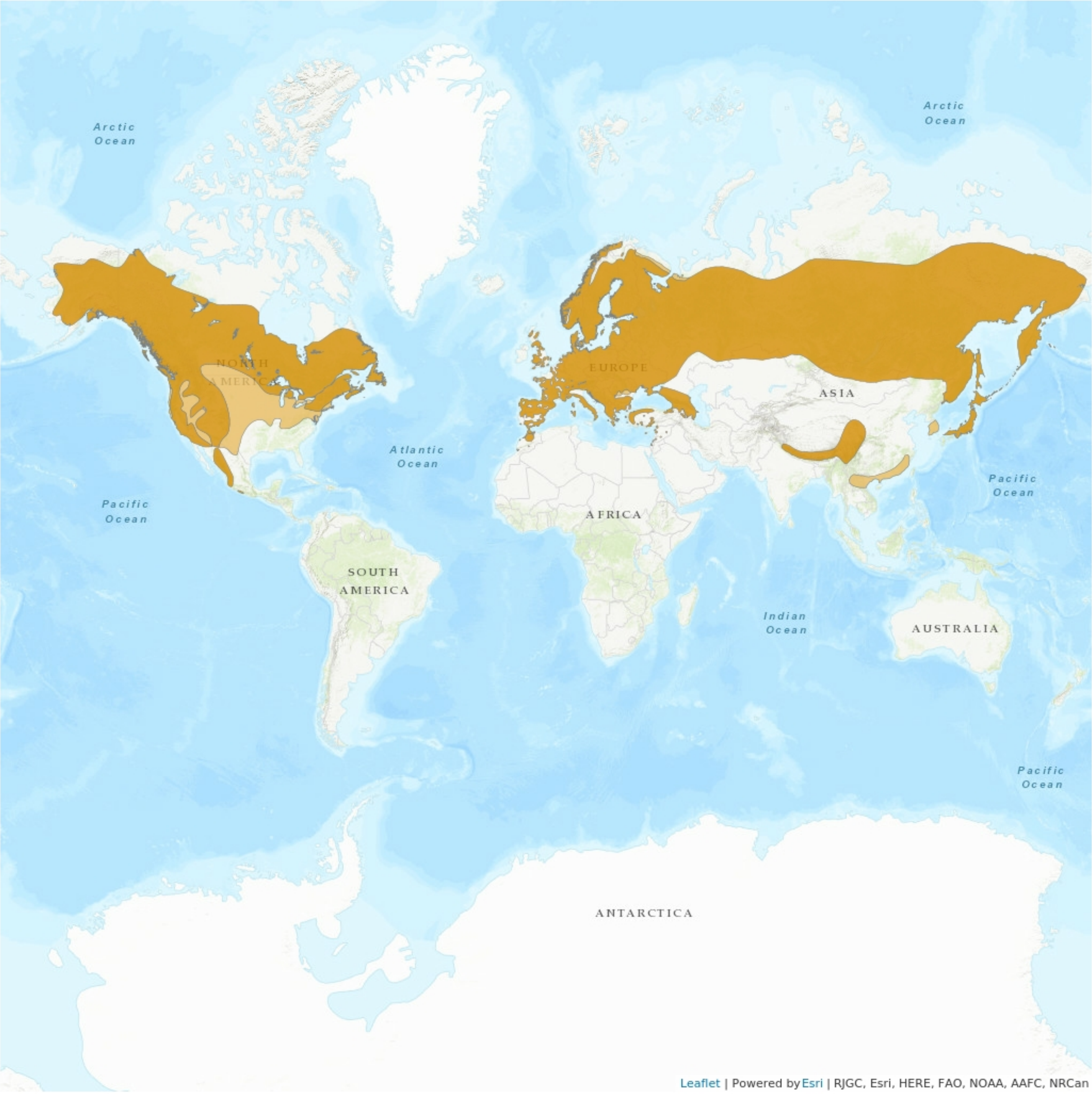
In Canada, mountain goat habitat, along with more than 3,500 goats, are protected in eight National Parks (Banff, Glacier, Jasper, Kootenay, Nahanni, Revelstoke, Waterton and Yoho), Kluane National Park Reserve, and in Kluane Wildlife Sanctuary. Numerous provincial parks and wildlife reserves throughout western and northern Canada provide additional varying levels of protection. Limited hunting by aboriginal people is permitted in some northern national parks and wildlife sanctuaries, and licensed hunting is permitted in many provincial parks. Outside protected areas, goats are legally hunted under strict controls issued by provincial or territorial government agencies. Harvests are set annually for each population. In British Columbia, for example, harvest rates vary between populations and range from 0.4 to 94% (BC Mountain Goat management Plan 2010) with about 600 goats shot by resident and non-residents each year. Aboriginal harvest is unknown. In Yukon, sport harvests are much lower, varying between three and 15 animals per year, with the aboriginal harvest estimated to be zero. Transplants have been used to re-introduce mountain goats into many areas of their former range. Habitat management continues to play a key role in its conservation, and developments are subject to environmental screening processes on public land. Conservation measures proposed for Canada: 1) Determine the species' requirements for mature forests on steep slopes in coastal mountain ranges that are used as winter habitat in British Columbia (Hebert and Turnbull, 1977, Fox *et al*. 1989). Several coastal populations will be affected by current and future timber harvest operations. Ideally, much or most of this habitat should be preserved. 2) Obtain more accurate population inventories in all regions of Canada to allow more detailed management plans to be developed. 3) Encourage research on the possible effects of climate change on habitat trends, nutritional quality and snow cover.

In the United States, primary conservation measures have included habitat protection, introductions and re-introductions, and harvest regulation. Eight state wildlife management departments have transplanted mountain goats from native ranges in Canada and the United States. Six of these states did not have native populations. Many transplanted populations were established with only 10 to 15 founder animals. Goats are harvested in nine states under conservative regulations of the wildlife departments which monitor populations. The mountain goat occurs in nine federal protected areas: Alaska: Glacier Bay, Kenai Fjords, and Wrangell - St. Elias National Parks; and Kenai National Wildlife Refuge; Montana: Glacier National Park; South Dakota: Mount Rushmore National Monument; Washington: North Cascades, and Mount Rainier National Parks. However, most herds are in national forests including many wilderness areas. One state, Colorado, uses two hunting licenses in an auction and a raffle to raise funds for these activities.

Credits

Assessor(s):	Festa-Bianchet, M.
Reviewer(s):	Coté, S. & Hamel, S.

Distribution Map - **Northern Goshawk**



Legend

🟧 EXTANT (RESIDENT)

🟨 EXTANT (NON-BREEDING)

Compiled by:

BirdLife International and Handbook of the Birds of the World 2021

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

Population

The European population is estimated at 234,000-380,000 mature individuals (BirdLife International In prep.). Europe forms approximately 26% of the global range, so a very preliminary estimate of global population size is 900,000-1,460,000, although further validation of this estimate is needed. The population in Canada and the USA is estimated at 210,000 mature individuals (Partners in Flight 2020). It is placed in the band 1,000,000-2,499,999 mature individuals.

Trend Justification

Globally, the population trend is considered unknown. This species has had stable population trends over the last 40 years in North America (data from Breeding Bird Survey and/or Christmas Bird Count: Butcher and Niven 2007). In Europe the population size is estimated to be decreasing by 59% in 24 years (three generations). Declines have been recorded recently in the east Netherlands (Rutz *et al.* 2006), locally in Germany (Gedeon *et al.* 2015), in Denmark (-8% per year from 2005-2014; Nyegaard *et al.* 2015), and in Finland (Valkama *et al.* 2011).

Current Population Trend: Unknown

Habitat and Ecology (see Appendix for additional information)

Behaviour The species is mainly resident, but its northernmost populations in North America, Scandinavia and Russia migrate south between September and November, returning in March and April (del Hoyo *et al.* 1994, Snow and Perrins 1998). Soaring flight is used frequently (Snow and Perrins 1998). It is always seen singly or in pairs (Ferguson-Lees and Christie 2001). **Habitat** It inhabits mature woodland, particularly coniferous, but also deciduous or mixed, preferring areas near clearings and the forest edge (del Hoyo *et al.* 1994). **Diet** Small birds and mammals make up the vast majority of its diet, with grouse, pheasants and partridges being especially important in boreal zones (del Hoyo *et al.* 1994). **Breeding site** Nests are built on the forks or branches of large trees (del Hoyo *et al.* 1994), typically in areas with high (60-90%) canopy closure (Squires *et al.* 2020). **Management information** The species's optimal habitat appears to be areas of farmland interspersed with mature forest; afforestation has improved its status across parts of its range (del Hoyo *et al.* 1994).

Systems: Terrestrial

Use and Trade

The species is a popular choice for falconry. The impact of falconry on wild populations is unknown, but is thought to be minimal. CITES data on reported trade of wild-caught live individuals from 1995-2019 show that Russia is the largest exporter (1257 birds exported), followed by Uzbekistan (504 birds) and Ukraine (82 birds). There has been no reported trade in wild-caught live individuals since 2015. The majority (84%) of exports during this period were reported to be for commercial purposes.

Threats (see Appendix for additional information)

Significant declines in Europe in the 19th-20th centuries are thought to have resulted from persecution and deforestation, with later declines in the 1950s-1960s a result of poisoning from pesticides and heavy metals. Studies on Northern Goshawk nestlings in Norway show that persistent organic pollutants are still present in plasma, preen oil and feathers, suggesting recent and continuous exposure (Briels *et al.* 2019). Persecution continues to be a threat, as is nest robbing for falconry (Orta and Marks 2014). It is also highly vulnerable to the impacts of potential wind farm developments (Strix 2012). Timber harvest is a primary threat to nesting populations, with nests and nest stands regularly destroyed or by logging operations, but the impact on breeding success is unknown (Squires *et al*. 2020, Byholm *et al.* 2020). Disturbance caused by logging activity near nests can also cause nest failure (Boal and Mannan 1994). In Alaska, U.S.A, clear-cut, even-aged, short-rotation forest management reduces habitat quality for the species, as does removal of old growth forest patches (Iverson *et al*. 1996). Increasing frequency and intensity of wildfires in western North America are causing a reduction in suitable roosting and foraging sites (Blakey *et al.* 2020). Declines in some European countries are associated with reduced food supply following intensification of farming and forestry and increased competition for nest sites from Eagle Owls *Bubo bubo* (Rutz *et al.* 2006).

Conservation Actions (see Appendix for additional information)

Conservation actions underway

The species is listed on CITES Appendix II, CMS Appendix II, Raptors MoU Category 3, and Bern Convention Appendix II. Fourteen European countries currently have breeding population monitoring schemes in place for this species (representing 33% of the countries in its European breeding range) (Derlink *et al.* 2018).

Conservation actions proposed

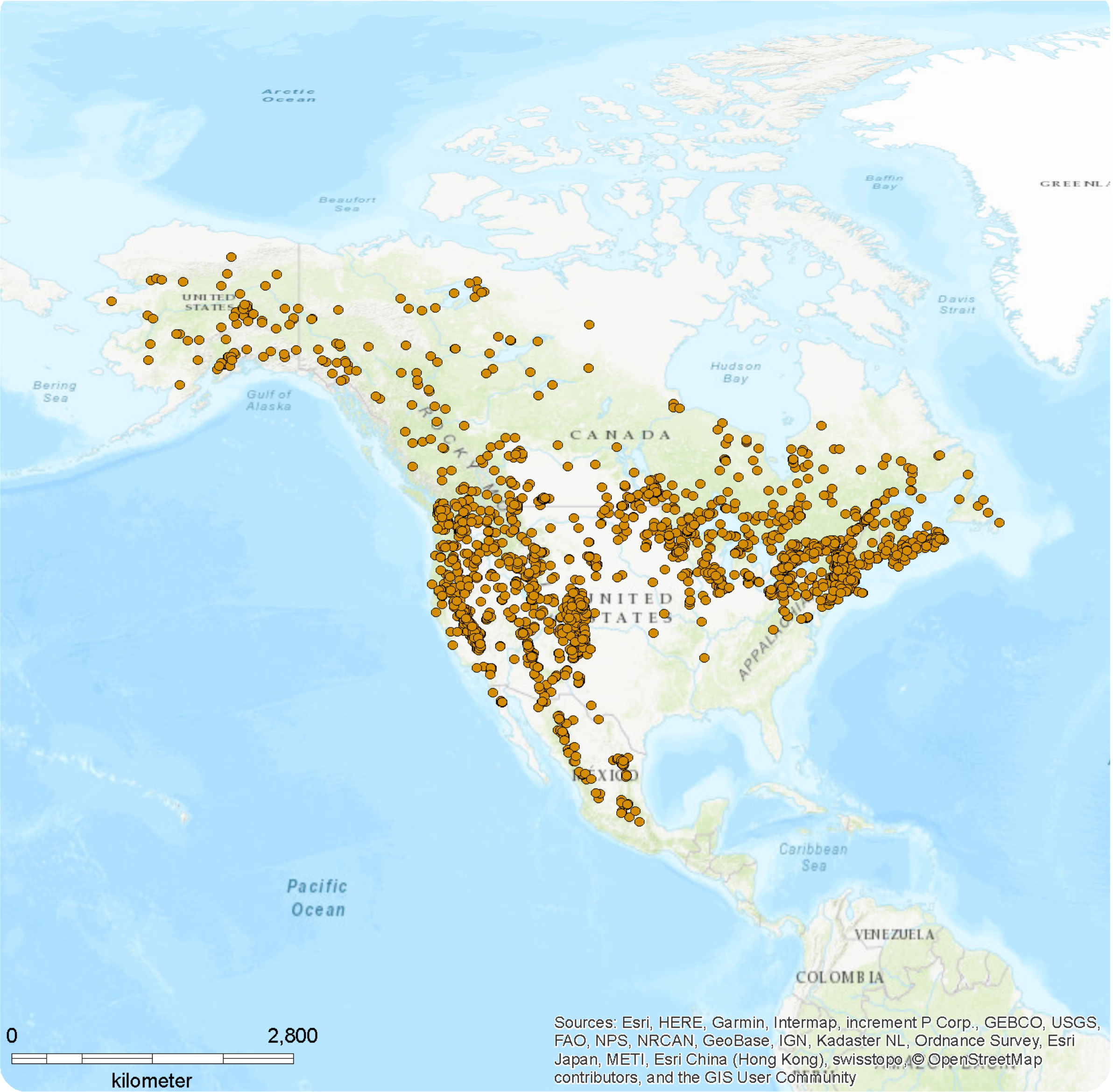
Further research is needed into habitat preferences, wintering biology, dispersal capabilities and the effect of changes in landscape pattern on foraging behaviour and population viability. Population trends are poorly understood as migration and roadside counts are limited due to low detection and irruptive movements (Squires *et al.* 2020). Other monitoring procedures need to be developed and implemented to determine population trends.

Credits

Assessor(s):	BirdLife International
Reviewer(s):	Clark, J.
Contributor(s):	Butchart, S., Ekstrom, J., Khwaja, N., Ashpole, J & Harding, M.
Facilitator(s) and Compiler(s):	Haskell, L.
Partner(s) and Institution(s):	BirdLife International
Authority/Authorities:	IUCN SSC Bird Red List Authority (BirdLife International)

Distribution Map - Quaking Aspen

Populus tremuloides



Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, © OpenStreetMap contributors, and the GIS User Community

Range

● Extant (resident)

Compiled by:

BGCI

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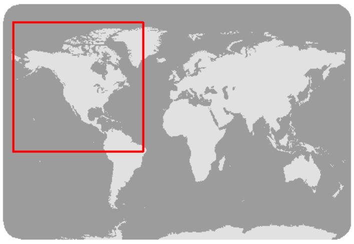


Population

The population of the species is at its most dense in the North of its range for example in parts of Canada, Alaska, Wisconsin and Michigan. The density of the species is lower and more patchy in parts of Mexico and the southern US states. The species is common and frequent in most North American forest types, it is listed as a dominant species in over 100 habitats (Howard 1996).
Stands of this tree are often clonal. The species is dioecious.

Current Population Trend: Stable

Habitat and Ecology (see Appendix for additional information)

This species of large tree can grow to between five and thirty metres in height. The species occurs in variety of forest and woodland habitats, where it is often common and can be the dominant species. It is found in Spruce fir (*Picea-Abies* spp.) forest and mixed conifer forest as well as many other types. It grows on the Great Plains, valleys of the Pacific Northwest and the Rocky Mountains. The species shows preference to the mid to upper riparian zone. It is associated within many shrub species. Due to the species abundance, the habitat it provides is essential to the breeding, foraging and resting of many animal species especially as it can be browsed throughout the year (Howard 1996). The species re sprouts quickly after fire and in general is a highly adaptive tree. However, it is not shade tolerant.
The species mainly reproduces through suckering and clonal growth from one root system.

Systems: Terrestrial

Use and Trade

This is an important timber species. Wood is most often used for particle board and pulp. It may also be used for lumber to produce packing boxes and creates, as well as furniture. It is more frequently used in the eastern United States.

This species may also be utilised for restoration and rehabilitation of land as it has high adaptability making it suitable to grow on many sites. It has been use to restore riparian habitats, strip mined sites and mine spoils. Native Americans also use the species as a food source by stripping and grinding the bark. The catkins may also be eaten (Nesom 2008).

Threats (see Appendix for additional information)

In some places the species is being out competed by conifers and the species is struggling to persist in some areas (NatureServe 2018). The species can be over browsed in the west of its range, which slows down regeneration at some sites. The species has a thin bark making it susceptible to wood decay and some pests and diseases (such as the polar borer beetle and forest tent caterpillar), though this is not a major threat to the species (Nesom 2008).

Conservation Actions (see Appendix for additional information)

This species is known from at least 90 *ex situ* collections (BGCI 2018). It occurs in protected areas across its broad range. In North America the species is assessed as G5- Secure on NatureServe (2018).

Credits

Distribution Map - Seagrass

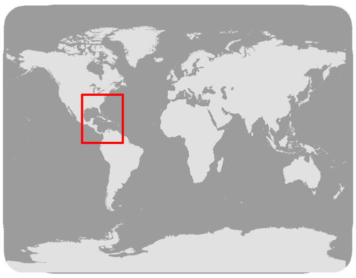
Syringodium filiforme

NE DD **LC** LEAST CONCERN NT VU EN CR EW EX

Range

🟨 Extant (resident)

Compiled by:
International Union for
Conservation of Nature (IUCN)



© The IUCN Red List of Threatened Species: Syringodium filiforme – published in 2010.
http://dx.doi.org/10.2305/IUCN.UK.2010-3.RLTS.T173378A7003203.en

Population

Syringodium filiforme is abundant and the population is thought to be stable throughout most of its range. Locally, this seagrass can be a major habitat forming species.

According to the Global Seagrass Trajectories Database, (T.J.B. Carruthers pers. comm. 2007) there are 13 published studies that monitored this species over time, and of these, 11 had no change and two showed increased coverage (all areal extent, biomass, or cover). Global average maximum biomass is estimated to be 368 g dw/m² above ground (from six observations) and 451 g dw/m² below ground (from four observations) (Duarte and Chiscano 1999). In Bermuda, out of 55 sites sampled 59% showed presence of this species. Of these, 22% had greater than 320 shoots/m² (Murdoch *et al*. 2004). There were wide scale decreases in abundance throughout Florida Bay from about 83.3 shoots/m² in 1984 to about 5.6 shoots/m² in 1994 with an 88% reduction in average dry weight density. The reduced abundance at that time was most likely due to increased light attenuation due to die-off of *Thalassia testudinum* (Hall *et al*. 1999).
Current Population Trend: Stable

Habitat and Ecology (see Appendix for additional information)

Syringodium filiforme is typically found on sand to mud bottoms down to at least 20 m, but in transparent waters this species can occur at deeper depths (Kenworthy and Fonseca 1996).This is locally a major habitat forming species. It often grows intermixed with *Thalassia testudinum* and/or *Halodule wrightii*. For example, in Cuba, it is found at a maximum depth of 16.5 m with biomass of 3.5 g/m². In the Caribbean, it usually grows intermixed with *Thalassia testudinum*, but also grows in mono-specific areas, beds or patches from the upper sublittoral down to more that 20 m (Green and Short 2003).

This species does not grow in brackish areas (Zieman 1982, UNESCO 1998, Hemminga and Duarte 2000, Green and Short 2003, Larkum *et al*. 2006), and it is absent in areas of poor water quality (Virnstein 1995). A large portion of the biomass grows below ground and below ground biomass is estimated at 50–60% of total biomass (Zieman, van Tussenbroek, Short, pers comm. 2007). This species has a high seed set from seed banks. Little is known about seed and seedling survival (van Tussenbroek pers comm. 2007).

Syringodium filiforme is heavily grazed by parrotfish in back reef areas and is an important food source for manatees. Other species grazing on this seagrass species are surgeonfish, sea urchins and perhaps pinfish. Other grazers, e.g., the queen conch, eat the epiphytic algae on the seagrass leaves (Zieman 1982).

Systems: Marine

Threats (see Appendix for additional information)

Threats affecting *Syringodium filiforme* are eutrophication and sedimentation. This species does not grow well in low quality water and needs good light.

In Florida, this species is locally affected by sewage pollution from expanded residential and hotel development, and marina and boat usage. It is also incidentally damaged from boat traffic. In the

Yucatan Peninsula, this species can be affected locally by trawling, eutrophication, and port development. Coastal developments and pollution from land-based sources, eutrophication (sewage and agricultural fertilizers) are local threats in the Caribbean region.

Conservation Actions (see Appendix for additional information)

This species occurs in a number of marine protected areas throughout its range. In the Caribbean for example, *Syringoium filiforme* is included in the 24 fully managed marine protected areas. Currently, a seagrass management plan is being developed in Bermuda (S. Sarkis pers. comm. 2007).

Credits

Assessor(s):	Short, F.T., Carruthers, T.J.R., van Tussenbroek, B. & Zieman, J.
Reviewer(s):	Livingstone, S., Harwell, H. & Carpenter, K.E.