

6. Individual Species Accounts—Western Atlantic

6.1 Genus *Agaricia* (Family Agariciidae)

6.1.1 *Agaricia lamarcki* Milne Edwards and Haime, 1851



Figure 6.1.1. *Agaricia lamarcki* photos copied from Veron and Stafford-Smith (2002).

Characteristics

Agaricia lamarcki has flat, unifacial or encrusting platy colonies that are commonly arranged in whorls. Corallites are in concentric valleys with centers that are widely spaced. Septo-costae clearly alternate in size. Colonies are brown in color, usually with pale margins. Mouths are characteristically white and star-shaped (Veron, 2000).

Taxonomy

Taxonomic issues: None. *Agaricia lamarcki* is similar to *Agaricia grahamae*, which lacks white mouths and has evenly sized septo-costae (Veron, 2000).

Family: Agariciidae.

Evolutionary and geologic history: *Agaricia lamarcki* is fairly common in recent fossil assemblages. However, it has not been identified from the late Pleistocene in fossil assemblages in the Cayman Islands (Hunter and Jones, 1996).

Global Distribution

The range of *Agaricia lamarcki* is restricted to the west Atlantic where it is found throughout the Caribbean; however, it is not known from Bermuda (IUCN, 2010).



Figure 6.1.2. *Agaricia lamarcki* distribution from IUCN copied from <http://www.iucnredlist.org>.



Figure 6.1.3. *Agaricia lamarcki* distribution copied from Veron and Stafford-Smith (2002).

U.S. Distribution

According to both the IUCN Species Account and the CITES database, *Agaricia lamarcki* is found in Florida, Puerto Rico, the U.S. Virgin Islands, and at Flower Garden Banks (IUCN, 2010).

A search of published and unpublished records of occurrence in U.S. waters indicates that *Agaricia lamarcki* has been reported in Florida (Goldberg, 1973), Puerto Rico (Acevedo et al., 1989; Garcia-Sais, 2010; Morelock et al., 2001), and the Virgin Islands (Rogers et al., 1984; Smith et al., 2010). Bright (1984) identified *Agaricia* at Flower Garden Banks only to genus, although it has been reported in low abundance at the site (Caldow et al., 2009).

Within federally protected waters, *Agaricia lamarcki* has been recorded from the following areas:

- Flower Garden Banks National Marine Sanctuary
- Florida Keys National Marine Sanctuary
- Biscayne National Park
- Dry Tortugas National Park
- Virgin Islands National Park/Monument
- Navassa Island National Wildlife Refuge
- Buck Island National Monument

Habitat

Habitat: *Agaricia lamarcki* is common in areas with reduced light or at depth (Acevedo et al., 1989). It can occur in shallow reef environments (Veron, 2000). It also inhabits reef slopes and walls and can be one of the most abundant corals on deep reefs (Humann, 1993).

Depth range: *Agaricia lamarcki* has been reported in water depths ranging from 10 to 76 m (Carpenter et al., 2008; Ghiold and Smith, 1990) and 3 to 50 m (Humann, 1993). Although *Agaricia lamarcki* can rarely be found in shaded areas in shallow waters, it primarily occurs at deeper depths. The IUCN Red List review emphasized a need for additional information on the population status and recovery potential for this species in deeper waters (IUCN, 2010). *Agaricia lamarcki* has often been found on mesophotic reefs in Curaçao, Florida, Jamaica, and the U.S. Virgin Islands (Ghiold and Smith, 1990). However, coral specimens collected on a recent mesophotic coral cruise at Pulley Ridge, Florida, suggest that corals, such as *Agaricia*, that appear live from video images may actually be covered with algae rather than live coral tissue (J. Voss, Florida Atlantic Univ., Fort Pierce, FL. pers. comm., August 2010).

Abundance

Agaricia lamarcki has been reported to be common (Veron, 2000). On reefs at 30–40 m depths in the Netherlands Antilles, *Agaricia lamarcki* has increased (Bak and Nieuwland, 1995) or shown no decline in abundance from 1973 to 1992 (Bak et al., 2005), even though other non-agariciid corals on the same deep reefs have decreased. It is not known whether this relative stability at depth holds across the full range of the species.

Life History

The specific reproductive strategy of *Agaricia lamarcki* is presently unknown, but its congeners are primarily gonochoric brooders (Delvoye, 1988; Van Moorsel, 1983). The larvae have been reported to primarily settle at relatively deep water depths (26–37 m), although the species has been found in shallow water (Bak and Engel, 1979). Congeneric larvae are known to use chemical cues from crustose coralline algae to mediate settlement (Morse et al., 1988). The species has low recruitment rates—as an example, only 1 of 1074 *Agaricia* recruits in a survey at the Flower Garden Banks may have been *Agaricia lamarcki* (Shearer and Coffroth, 2006). Net sexual recruitment over a decade can be negligible, with reproduction primarily via fission (Hughes and Jackson, 1985). It is a relatively long-lived species, with a half-life of 17 years (Hughes, 1996) and some colonies living more than a century (Hughes and Jackson, 1985).

Agaricia lamarcki deposits a relatively dense skeleton (Hughes, 1987) and is reported to be moderately susceptible to physical breakage during severe storms (Aronson et al., 1993). Maximum size for *Agaricia lamarcki* is up to ~ 2 m in diameter (Humann, 1993), with radial growth rates in Jamaica ranging from 0 to 1.4 cm per year (average growth rate of ~ 5 mm per year), but growing a bit more slowly at depths greater than 20 m (Hughes and Jackson, 1985). Respiration rates have been reported to be relatively high (~ 3.5 $\mu\text{L O}_2$ per mg per hr) compared to other shallower species, which may be related to zooxanthellae density (Davies, 1980). Photosynthesis by *Agaricia lamarcki* zooxanthellae is sufficient to exceed the coral's metabolic needs, even at depths in excess of 30 m (Porter et al., 1989).

Mortality of *Agaricia lamarcki* is size-specific (range 10%–25%), with high (22%–90%) rates of partial mortality (Hughes and Jackson, 1985). Partial mortality can be induced by interactions with algae (Nugues and Bak, 2006). Sponges can induce partial mortality, as well as cause sublethal stresses such as declines in zooxanthellae concentration, pigment concentration, and tissue condition (Porter and Targett, 1988).

In the Virgin Islands (Rogers et al., 1984) and Curaçao (Bak and Luckhurst, 1980), the overall life history characteristics of *Agaricia lamarcki* have been reported to be roughly parallel to those of *Montastraea annularis*—that is, based on low overall recruitment rates, high survival, and high partial mortality. However, in Jamaica *Agaricia lamarcki* had faster growth, higher recruitment, and lower mortality rates than *Montastraea annularis* at the same site and depth (Hughes and Jackson, 1985).

Threats

Thermal stress: *Agaricia lamarcki* has been reported to be susceptible to bleaching at elevated temperatures (Ghiold and Smith, 1990), via direct loss of zooxanthellae as well as decreased pigment content (Porter et al., 1989). In laboratory studies in Jamaica, *Agaricia lamarcki* tolerated temperatures up to 32°C (Fitt and Warner, 1995), but had virtually complete disruption of photosynthesis occur at 32°C–34°C (Warner et al., 1996). Cold stress has also produced bleaching (Bak et al., 2005). Although bleaching can often be extensive, it may not induce mortality in *Agaricia lamarcki* (Aronson and Precht, 2000; Aronson et al., 1998; Porter et al., 1989).

Acidification: No specific research has addressed the effects of acidification on the genus *Agaricia*. However, most corals studied have shown negative relationships between acidification and growth (Table 3.2.2), and acidification is likely to contribute to reef destruction in the future (Hoegh-Guldberg et al., 2007; Silverman et al., 2009). While ocean acidification has not been demonstrated to have caused appreciable declines in coral populations so far, the BRT considers it to be a significant threat to corals by 2100 (Albright et al., 2010; Hoegh-Guldberg et al., 2007; Langdon and Atkinson, 2005; Manzello, 2010; Silverman et al., 2009).

Disease: *Agaricia lamarcki* was not observed to suffer disease in the Florida Keys in 1996–1998 (Porter et al., 2001), although prior observations in Florida showed that the species may suffer from white plague (Richardson, 1998). *Agaricia lamarcki* also has been vulnerable to white plague disease in Colombia (Garzon-Ferreira et al., 2001) and St. Lucia (Nugues, 2002). Ciliate infections have been documented in *Agaricia lamarcki* (Croquer et al., 2006), and tumors may also affect this species (UNEP, 2010). The ecological and population impacts of disease have not been established for *Agaricia lamarcki*.

Predation: Predation effects on *Agaricia lamarcki* are unknown.

Land-based sources of pollution (LBSP): The effects of LBSP on the genus *Agaricia* are largely unknown. LBSP-related stresses (nutrients, sediment, toxins, and salinity) often act in concert rather than individually and are influenced by other biological (e.g., herbivory) and hydrological factors. Collectively, LBSP stresses are unlikely to produce extinction at a global scale; however, they may pose significant threats at local scales and reduce the resilience of corals to bleaching (Carilli et al., 2009a; Wooldridge, 2009b).

Agaricia sp. typically have small calices and are not efficient sediment rejecters (Hubbard and Pocock, 1972). *Agaricia lamarcki*'s platy morphology could make it sediment-susceptible. Vertical plates of *Agaricia* shed more sediment than horizontally-oriented ones (Bak and Elgershuizen, 1976), and fine sediment suspended in hurricanes can cause much higher mortality in platy corals than hemispherical or non-flat morphologies (Bak, unpublished data; Bak et al., 2005).

Collection/Trade: Some corals in this genus are involved in international trade, especially *Agaricia agaricites* (CITES, 2010). However, only light trade has been recorded for *Agaricia lamarcki*. From 2000 to 2005, gross exports averaged fewer than 10 pieces of coral (CITES, 2010).

Risk Assessment

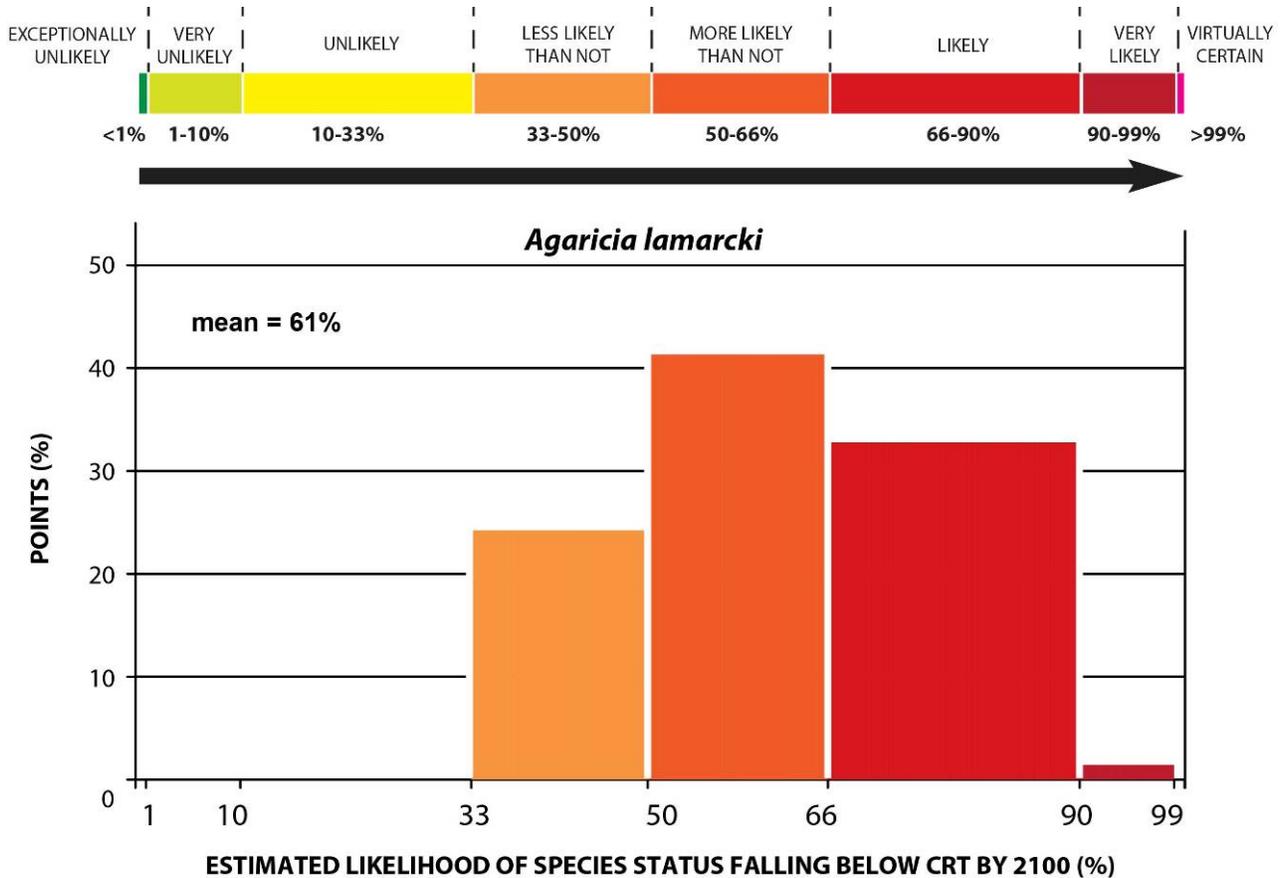


Figure 6.1.4. Distribution of points to estimate the likelihood that the status of *Agaricia lamarcki* falls below the Critical Risk Threshold (the species is of such low abundance or so spatially fragmented or at such reduced diversity that extinction is extremely likely) by 2100.

Factors that increase the extinction risk (higher likelihood of falling below the Critical Risk Threshold) for *Agaricia lamarcki* include the widespread decline in environmental conditions in the Caribbean and the potential losses of this species to disease. When bleaching occurs for this species, effects can be severe; the species also likely has limited sediment tolerance. A factor that reduces extinction risk (decrease the likelihood of falling below the Critical Risk Threshold) is that it occurs primarily at great depth, where disturbance events are less frequent. Despite low rates of sexual recruitment, the species is relatively persistent compared to other deep corals.

The overall likelihood that *Agaricia lamarcki* will fall below the Critical Risk Threshold by 2100 was estimated to be in the “more likely than not” risk category with a mean likelihood of 61% and a standard error (SE) of 6% (Fig. 6.1.4). This SE was calculated by taking the standard deviation of the seven mean voting scores of the BRT members and shows the coherence among the BRT. This is one of the lowest SE of the mean values for any species, showing a greater degree of confidence in the *Agaricia lamarcki* status estimate than for most other species considered. However, the overall range of votes was still fairly large (33%–99%; Fig. 6.1.4) with a moderate average range of likelihood estimates of the seven BRT voters (55%).

6.2 Genus *Mycetophyllia* (Family Mussidae)

6.2.1 *Mycetophyllia ferox* Wells, 1973

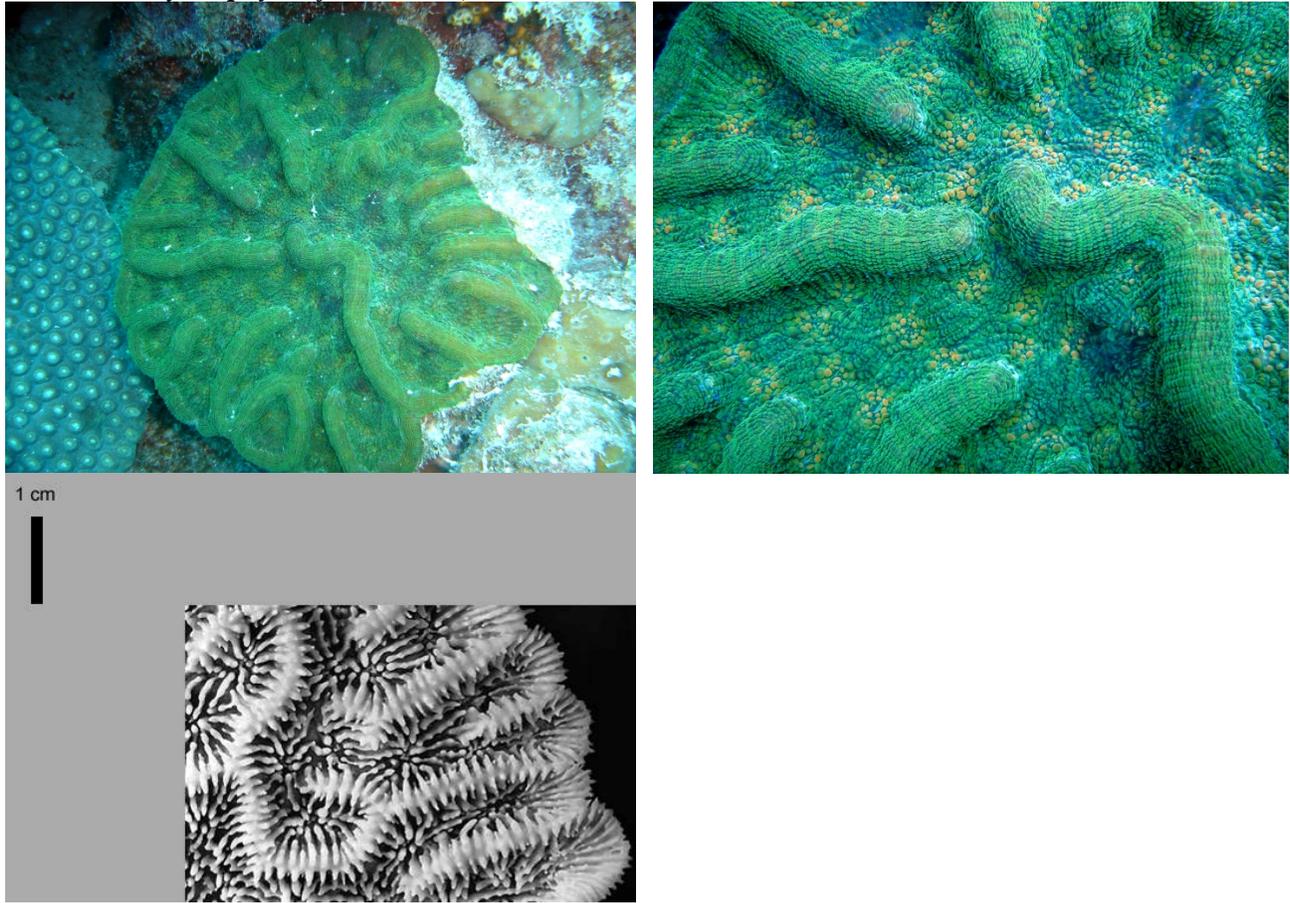


Figure 6.2.1. *Mycetophyllia ferox* photos from National Park Service and corallite plan from Veron and Stafford-Smith (2002).

Characteristics

Mycetophyllia ferox consists of encrusting laminar plates. Colonies are thin, weakly attached plates with interconnecting, slightly sinuous narrow valleys. Corallite centers are usually in single rows. Columellae are rudimentary or absent. Colonies are most commonly greys and browns in color with valleys and walls of contrasting colors (Veron, 2000). Maximum colony size is 50 cm (Veron, 2000).

Taxonomy

Taxonomic issues: None. *Mycetophyllia ferox* is similar to *Mycetophyllia danaana*, which has longer, wider, and more widely spaced valleys (Veron, 2000).

Family: Mussidae.

Evolutionary and geologic history: *Mycetophyllia ferox* has been dated to at least the late Pleistocene in fossil records in Grand Cayman (Hunter and Jones, 1996).

Global Distribution

The range of *Mycetophyllia ferox* is restricted to the west Atlantic. There it has been reported to occur throughout most of the Caribbean, including the Bahamas, but it is not present in the Flower Garden Banks or around the waters of Bermuda. E-mail correspondence with S. dePutron (Bermuda Institute of Ocean Sciences, St. George's. pers. comm.,

May 2010) and T. Murdoch (Bermuda Zoological Society, Flatts. pers. comm., May 2010) confirmed the absence of *Mycetophyllia ferox* in Bermuda.



Figure 6.2.2. *Mycetophyllia ferox* distribution from IUCN copied from <http://www.iucnredlist.org>.



Figure 6.2.3. *Mycetophyllia ferox* distribution from Veron and Stafford-Smith (2002).

U.S. Distribution

According to both the IUCN Species Account and the CITES species database, *Mycetophyllia ferox* occurs throughout the U.S. waters of the western Atlantic but has not been reported from Flower Garden Banks (Hickerson et al., 2008).

Within federally protected waters, *Mycetophyllia ferox* has been recorded from the following areas:

- Dry Tortugas National Park
- Virgin Island National Park/Monument
- Florida Keys National Marine Sanctuary
- Navassa Island National Wildlife Refuge
- Biscayne National Park
- Buck Island Reef National Monument

Habitat

Habitat: *Mycetophyllia ferox* has been reported to occur in shallow reef environments (Veron, 2000).

Depth range: *Mycetophyllia ferox* has been reported in water depths ranging from 5 to 30 m (Carpenter et al., 2008).

Abundance

Mycetophyllia ferox is usually uncommon (Veron, 2000) or rare according to published and unpublished records, indicating that it constitutes < 0.1% species contribution (percent of all colonies censused) and occurs at densities < 0.8 colonies per 10 m² in Florida (Wagner et al., 2010) and at 0.8 colonies per 100 m transect in Puerto Rico sites sampled by the Atlantic and Gulf Rapid Reef Assessment (AGRR database online at <http://www.agrra.org>). Recent monitoring data (e.g., since 2000) from Florida (National Park Service permanent monitoring stations), La Parguera Puerto Rico, and St. Croix (USVI/NOAA Center for Coastal Monitoring and Assessment randomized monitoring stations) show

Mycetophyllia ferox cover to be consistently less than 1%, with occasional observations up to 2% and no apparent temporal trend (available online at http://www8.nos.noaa.gov/bioge_public/query_habitat.aspx).

Dustan (1977) suggests that *Mycetophyllia ferox* was much more abundant in the upper Florida Keys in the early mid-1970s (the methods are not well described for that study) than current observations, but that it was highly affected by disease. This could be interpreted as a substantial decline. Long-term CREMP monitoring data in Florida on species presence/absence from fixed sites (stations) show a dramatic decline; for 97 stations in the main Florida Keys, occurrence had declined from 20 stations in 1996 to 4 stations in 2009; in Dry Tortugas occurrence had declined from 8 out of 21 stations in 2004 to 3 stations in 2009 (R. Ruzicka and M. Colella, Florida Marine Research Institute, St. Petersburg, FL. pers. comm., Oct 2010).

Life History

Mycetophyllia ferox is hermaphroditic and a brooder. Egg size has been estimated in Puerto Rico to be 300 µm, and polyps produce 96 eggs per cycle on average (Szmant, 1986). Their larvae contain zooxanthellae that can supplement maternal provisioning with energy sources provided by their photosynthesis (Baird et al. 2009), i.e., they are autotrophic. Colony size at first reproduction is > 100 cm² (Szmant, 1986). Recruitment of this species appears to be very low, even in studies from the 1970s (Dustan, 1977, reported zero settlement).

Threats

Temperature stress: No bleached *Mycetophyllia ferox* colonies were observed in wide-scale surveys during the 2005 mass coral bleaching event in Florida (Wagner et al., 2010) or Barbados (Oxenford et al., 2008), although the number of colonies was small (two in Barbados; Oxenford et al., 2008).

Acidification: No specific research has addressed the effects of acidification on the genus *Mycetophyllia*. However, most corals studied have shown negative relationships between acidification and growth (Table 3.2.2), and acidification is likely to contribute to reef destruction in the future (Hoegh-Guldberg et al. 2007, Silverman et al. 2009). While ocean acidification has not been demonstrated to have caused appreciable declines in coral populations so far, the BRT considers it to be a significant threat to corals by 2100 (Albright et al., 2010; Hoegh-Guldberg et al., 2007; Langdon and Atkinson, 2005; Manzello, 2010; Silverman et al., 2009).

Disease: *Mycetophyllia ferox* has been reported to be susceptible to acute and subacute white plague and Dustan (1977) reported dramatic impacts from this disease to the population in the upper Florida Keys in the mid-1970s. He also reported that the rate of disease progression was positively correlated with water temperature and measured rates of disease progression up to 3 mm per day.

Predation: *Mycetophyllia ferox* has not been susceptible to predation (E. Peters, George Mason University, Fairfax, VI. pers. comm., July 2010).

Land-based sources of pollution: *Mycetophyllia ferox* appeared to be absent at fringing reef sites impacted by sewage pollution (Tomascik and Sander, 1987a). LBSP-related stresses (nutrients, sediment, toxins, and salinity) often act in concert rather than individually and are influenced by other biological (e.g., herbivory) and hydrological factors. Collectively, LBSP stresses are unlikely to produce extinction at a global scale; however, they may pose significant threats at local scales and reduce the resilience of corals to bleaching (Carilli et al., 2009a; Wooldridge, 2009b).

Collection/Trade: *Mycetophyllia ferox* is not reported to be an important species for trade. In 2000, 10 pieces of *Mycetophyllia ferox* were exported; only 2 in 2003; and 5 in 2007, according to CITES Trade Database, UNEP World Conservation Monitoring Centre, Cambridge, UK (CITES, 2010).

Risk Assessment

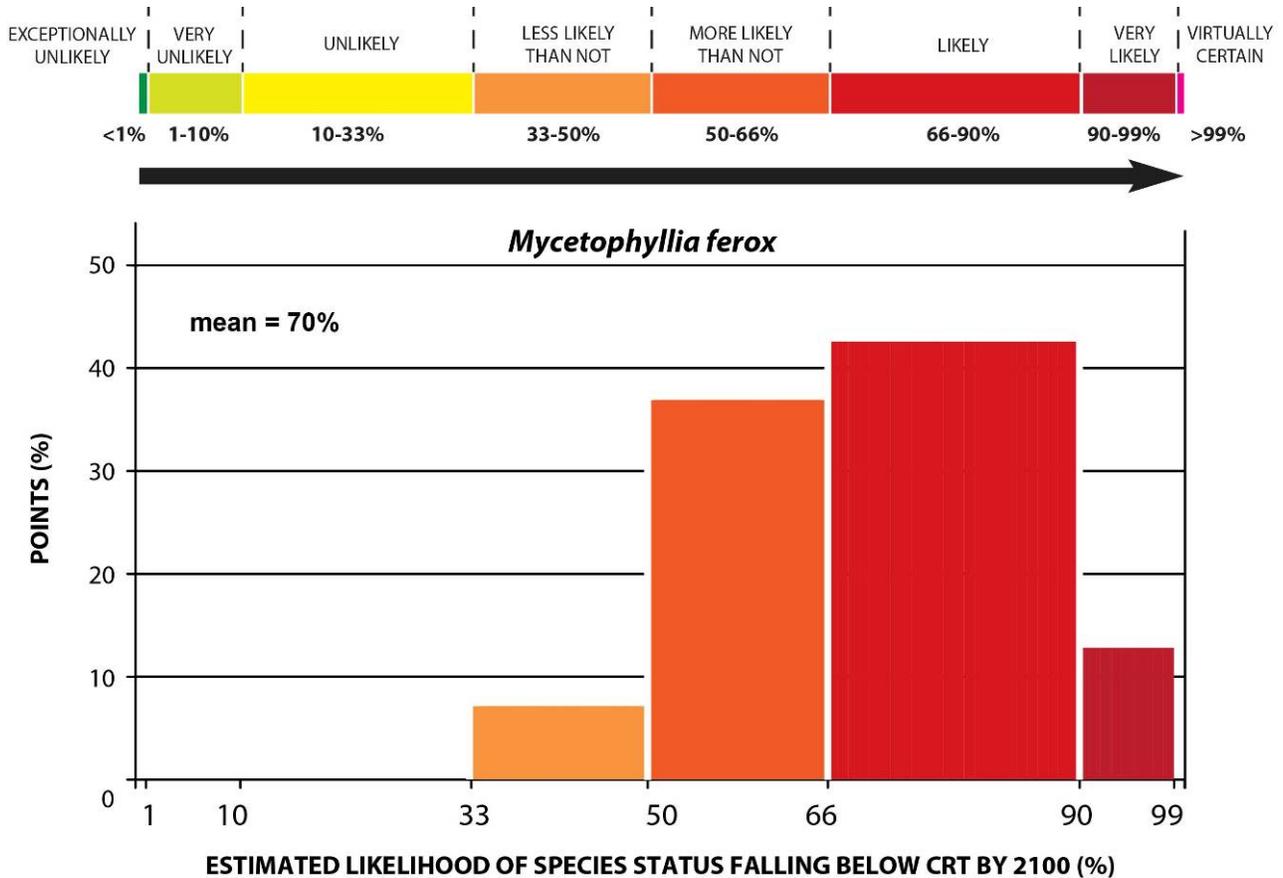


Figure 6.2.4. Distribution of points to estimate the likelihood that the status of *Mycetophyllia ferox* falls below the Critical Risk Threshold (the species is of such low abundance or so spatially fragmented or at such reduced diversity that extinction is extremely likely) by 2100.

Factors that increase the potential extinction risk (higher likelihood of falling below the Critical Risk Threshold) for *Mycetophyllia ferox* include disease, rare abundance, and observed declines in abundance. Limited available information suggests that this species suffered substantial population declines in the Florida Keys and elsewhere in recent decades, primarily as a result of coral disease, and these declines have made this species extremely rare.

The overall likelihood that *Mycetophyllia ferox* will fall below the Critical Risk Threshold by 2100 was estimated to be in the “likely” risk category with a mean likelihood of 70% and a standard error (SE) of 8% (Fig. 6.2.4). This SE was calculated by taking the standard deviation of the seven mean voting scores of the BRT members and shows the coherence among the BRT. The uncertainty of the BRT is reflected in the range of votes of 33%–99% (Fig. 6.2.4) and the average range of likelihood estimates of the seven BRT voters (50%). The range of votes reflects the uncertainty among BRT members based on recent monitoring data showing declines in the Florida Keys and elsewhere in the Caribbean.

6.3 Genus *Dendrogyra* (Family Meandrinidae)

6.3.1 *Dendrogyra cylindrus* Ehrenberg, 1834

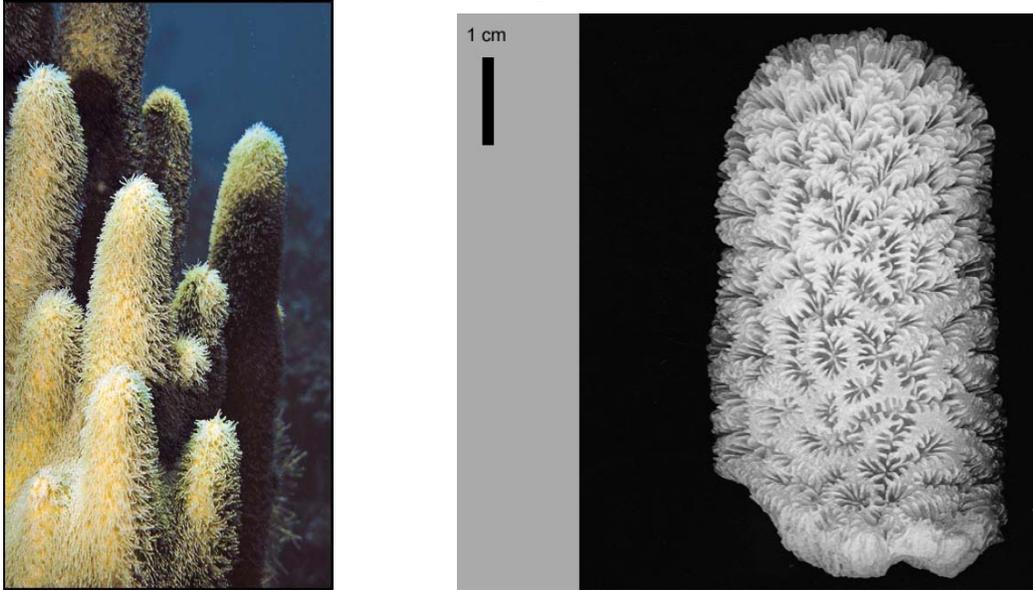


Figure 6.3.1. *Dendrogyra cylindrus* photos and corallite plan copied from Veron and Stafford-Smith (2002).



Figure 6.3.2. *Dendrogyra cylindrus* colony with rapidly progressing partial mortality characteristic of white plague disease. Photo: NOAA Southeast Fisheries Science Center.

Characteristics

Dendrogyra cylindrus colonies have encrusting bases on which cylindrical columns are developed that may reach 2 m in height. Valleys are meandroid. Septo-costae are thick, in two alternating orders; they do not join at the tops of valleys and thus leave a neat groove along the tops of walls. Tentacles remain extended during the day giving columns a furry appearance. Colonies are generally grey-brown in color (Veron, 2000).

Taxonomy

Taxonomic issues: None.

Family: Meandrinidae.

Evolutionary and geologic history: *Dendrogyra cylindrus* is reported to have appeared very recently in the fossil record (Edinger and Risk, 1995) following the Pliocene (~ 1.5 Million years ago [Ma]). *Dendrogyra cylindrus* is the

only species within its genus, perhaps posing greater evolutionary importance since extinction of this species would constitute extinction of a genus.

Global Distribution

Dendrogyra cylindrus is restricted to the west Atlantic where it is present throughout the greater Caribbean but is one of the Caribbean genera absent from the southwest Gulf of Mexico (Tunnell, 1988). A single known colony in Bermuda is reported to be in poor condition (T. Murdoch, Bermuda Zoological Society, Flatts, pers. comm., May 2010).



Figure 6.3.3. *Dendrogyra cylindrus* distribution from IUCN copied from <http://www.iucnredlist.org>.



Figure 6.3.4. *Dendrogyra cylindrus* distribution from Veron and Stafford-Smith (2002).

U.S. Distribution

Dendrogyra cylindrus has been reported in the waters of south Florida and the U.S. Caribbean but appears to be absent from the Flower Garden Banks. Within federally protected U.S. waters, the species has been recorded from the following areas:

- Florida Keys National Marine Sanctuary
- Navassa National Wildlife Refuge
- Dry Tortugas National Park
- Virgin Islands National Park/Monument
- Biscayne National Park NPS
- Buck Island National Monument

Habitat

Habitat: *Dendrogyra cylindrus* inhabits most reef environments (Veron, 2000), but in the Florida Keys it appears to be absent in nearshore hard bottoms, nearshore patch reefs, and backreef environments and more common on forereef spur-and-groove habitats (Chiappone, 2010).

Depth range: *Dendrogyra cylindrus* has been reported in water depths ranging from 2 to 25 m (Carpenter et al., 2008).

Abundance

Dendrogyra cylindrus is reported to be uncommon but conspicuous (Veron 2000) with isolated colonies scattered across a range of habitat types. Colonies are often known as landmarks by local divers. Overall colony density throughout south Florida was estimated to be ~ 0.6 colonies per 10 m² (Wagner et al. 2010). Overall colony density in Providencia, Columbia, was 172 (SE 177) colonies per km² (Acosta and Acevedo, 2006). *Dendrogyra cylindrus* is common in the geologic record of some Pleistocene reefs (Hunter and Jones, 1996), but it is likely that *Dendrogyra cylindrus* is a naturally rare species in modern times. Recent monitoring data (e.g., since 2000) from La Parguera, Puerto Rico, and St. Croix, USVI (NOAA Center for Coastal Monitoring and Assessment, randomized monitoring stations) have shown that *Dendrogyra cylindrus* cover was consistently less than 1% with individual observations up to 4% but with no apparent temporal trend, although trends would be difficult to detect with such low cover values (available online at http://www8.nos.noaa.gov/biogeopublic/query_habitat.aspx).

Life History

Dendrogyra cylindrus is characterized as a gonochoric spawner (Szmant, 1986), although no descriptions of its spawning or larval ecology have been made. The combination of gonochoric spawning reproductive mode with persistently low population densities poses somewhat of a paradox, since this combination is expected to yield very little potential for successful fertilization and, hence, larval supply. Indeed, no juveniles of this species were encountered from surveys of 566 sites in the Florida Keys during 1999–2009 (Chiappone, 2010), neither in larval settlement studies in the U.S. Virgin Islands in the early 1980s (Rogers et al., 1984), nor in juvenile surveys in the mid-1970s in the Netherlands Antilles (Bak and Engel, 1979). *Dendrogyra cylindrus* is effective in propagation by fragmentation, and rare aggregations of colonies (Hudson and Goodwin, 1997) likely result from this asexual reproductive mode following storms or other physical disturbances.

Annual growth rates of 12–20 mm per year in linear extension have been reported in the Florida Keys (Hudson and Goodwin, 1997), but growth rates of ~ 0.8 cm per year have been reported elsewhere in the Caribbean (Acosta and Acevedo, 2006; Hughes, 1987). Partial mortality rates have been size-specific but generally low (Acosta and Acevedo, 2006). Feeding clearance rates are low relative to most other Caribbean corals (Lewis, 1976), but *Dendrogyra cylindrus* has a relatively high photosynthetic rate and stable isotope values suggest it receives substantial amounts of photosynthetic products translocated from its zooxanthellae (Muscatine et al., 1989b).

Threats

Thermal stress: There are conflicting characterizations of bleaching susceptibility of *Dendrogyra cylindrus* in the literature. The species was bleaching-resistant during the 1983 mass bleaching event in Florida (Jaap, 1985). Characterizations of the 2005 mass bleaching event in southern Florida and in the U.S. Virgin Islands noted that no bleached *Dendrogyra cylindrus* colonies were observed (Clark et al., 2009; Wagner et al., 2010). In contrast, Oxenford et al. (2008) report that 100% of the 15 colonies they observed in Barbados during the 2005 mass bleaching event were bleached. Although bleaching of most coral species varies in time and space, understanding the susceptibility of *Dendrogyra cylindrus* is further confounded by the species' rarity and, hence, low sample size in any given survey.

Dendrogyra cylindrus is among the species that are known to be sensitive to cold shock in the Caribbean (Muscatine et al., 1991), potentially serving as a stress to this species in areas prone to cold winter temperatures such as the Florida Keys. *Dendrogyra cylindrus* hosts clade B zooxanthellae in Mexico (LaJeunesse, 2002), Belize, and Barbados (Finney et al., 2010). Zooxanthellae in clade B do not grow well at high temperatures (Kinzie et al., 2001), but in the field, corals with this clade may be relatively bleaching-resistant (McField, 1999). Experimental studies suggest clade B is more bleaching-resistant than clade C but less resistant than clade A (Warner et al., 2006).

Acidification: No specific research has addressed the effects of acidification on the genus *Dendrogyra*. However, most corals studied have shown negative relationships between acidification and growth (Table 3.2.2), and acidification is likely to contribute to reef destruction in the future (Hoegh-Guldberg et al. 2007, Silverman et al. 2009). While ocean acidification has not been demonstrated to have caused appreciable declines in coral populations so far, the BRT considers it to be a significant threat to corals by 2100 (Albright et al., 2010; Hoegh-Guldberg et al., 2007; Langdon and Atkinson, 2005; Manzello, 2010; Silverman et al., 2009).

Disease: *Dendrogyra cylindrus* colonies have been affected by black-band disease (Ward et al., 2006). More extensive impacts to these rare populations likely occur from white plague, which can cause rapid tissue loss (Miller et al., 2006b). The large colony size suggests that individual colonies are less likely to suffer complete mortality from a given disease exposure, but low colony density in this species suggests that even small degrees of mortality increase extinction risk.

Predation: The corallivorous fireworm, *Hermodice carunculata*, has been observed on diseased colonies of *Dendrogyra cylindrus* (Miller et al., 2006b), but, generally, predation is not observed to cause noticeable mortality on this species, despite its rarity.

Land-based sources of pollution (LBSP): Sediment stress is a complicated response; most sediment effects are negative (Fabricius, 2005; Rogers, 1990), although some corals are sediment-tolerant. Bak and Elgershuizen (1976) found that the rate of sand removal from *Dendrogyra cylindrus* tissues in laboratory conditions was intermediate among 19 Caribbean coral species tested. Along a eutrophication gradient in Barbados, *Dendrogyra cylindrus* was found at only a single site—one of those farthest removed from pollution (Tomascik and Sander, 1987a).

Overall, LBSP-related stresses (nutrients, sediment, toxins, and salinity) often act in concert rather than individually and are influenced by other biological (e.g., herbivory) and hydrological factors. Collectively, LBSP stresses are unlikely to produce extinction at a global scale; however, they may pose significant threats at local scales and reduce the resilience of corals to bleaching (Carilli et al., 2009a; Wooldridge, 2009b).

Collection/Trade: Overall trade reports (CITES database) indicate very low rates of international trade of *Dendrogyra cylindrus* (exception of anomalous report of 6000 pieces imported by Portugal from Mozambique in 1996). It is possible that historical curio collecting of *Dendrogyra cylindrus* may have significantly reduced populations off Florida (Colin, 1978).

Risk Assessment

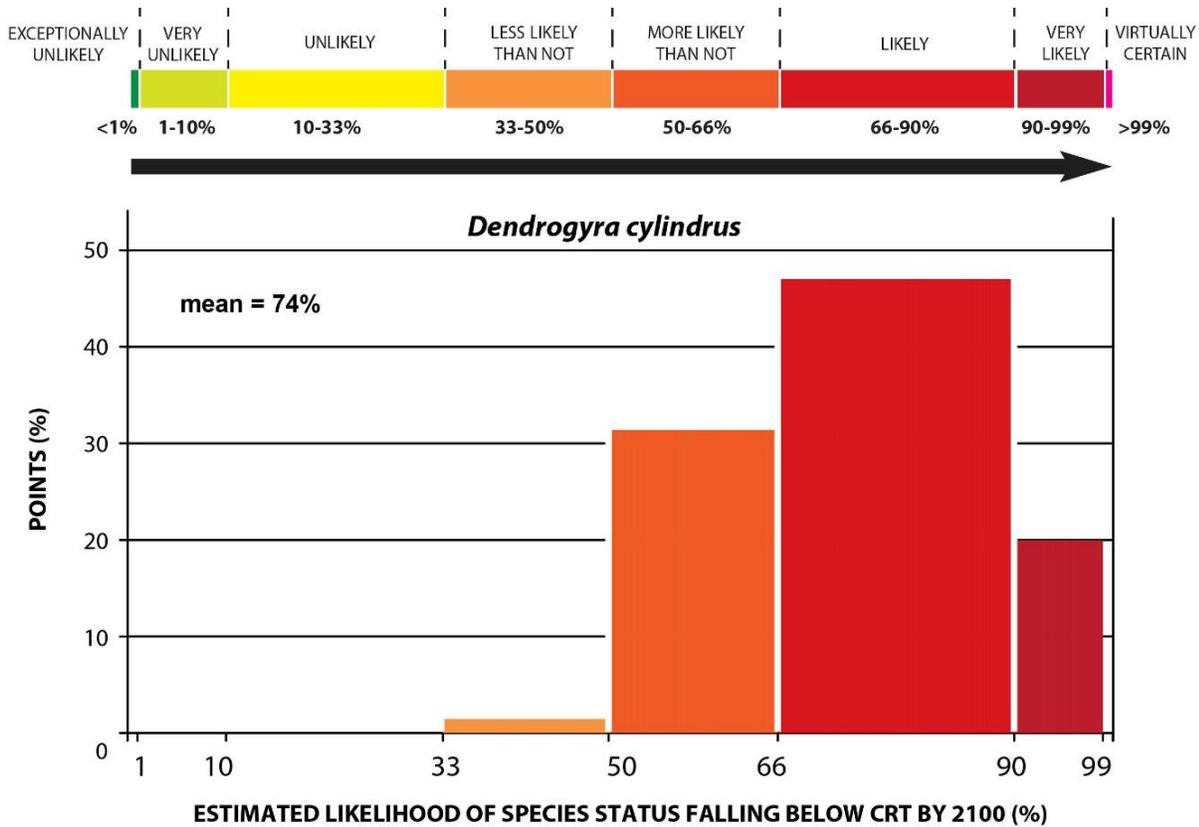


Figure 6.3.5. Distribution of points to estimate the likelihood that the status of *Dendrogyra cylindrus* falls below the Critical Risk Threshold (the species is of such low abundance or so spatially fragmented or at such reduced diversity that extinction is extremely likely) by 2100.

The most important factors influencing the relatively high extinction risk (higher likelihood of falling below the Critical Risk Threshold) of *Dendrogyra cylindrus* included the overall low population density and low population size combined with a gonochroic spawning reproductive mode, corresponding lack of observed sexual recruitment, and susceptibility to observed disease mortality. The BRT recognizes that, given the apparent naturally rare status of this species, some undescribed adaptations to low population density may exist in this species (particularly with regard to overcoming fertilization limitation between spawned gametes from gonochoric parent colonies that are at great distance from one another). Nonetheless, the pervasiveness of threats characterizing the Caribbean region was deemed to represent substantial extinction risk given this species' low population size.

The overall likelihood that *Dendrogyra cylindrus* will fall below the Critical Risk Threshold by 2100 was estimated to be in the "likely" risk category with a mean likelihood of 74% and a standard error (SE) of 6.6% (Fig. 6.3.5). This SE was calculated by taking the standard deviation of the seven mean voting scores of the BRT members and shows the degree of coherence among the BRT. The degree of uncertainty of the BRT is reflected in the range of votes of 33%–99% (Fig. 6.3.5) and the average range of likelihood estimates of the seven BRT voters (48.9%). The overall wide range of votes reflects the uncertainty among BRT members inherent in rigorously surveying a species that is historically rare.

6.4 Genus *Dichocoenia*

6.4.1 *Dichocoenia stokesi* Milne Edwards and Haime, 1848



Figure 6.4.1. *Dichocoenia stokesi* photos and corallite plan copied from Veron and Stafford-Smith (2002).

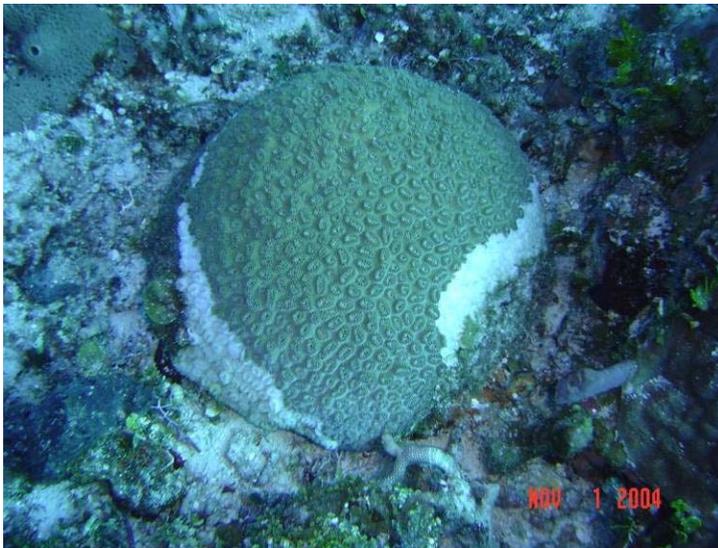


Figure 6.4.2. *Dichocoenia stokesi* colony with partial mortality characteristic of white plague disease. Photo from NOAA Southeast Fisheries Science Center.

Characteristics

Dichocoenia stokesi colonies are either massive and spherical or form thick, submassive plates (Veron, 2000). The corallites of this species are evenly spaced and either plocoid or ploc-meandroid, and the septocostae are usually in two neatly alternating orders (Veron, 2000). Although sometimes green, they are usually orange-brown with white septocostae.

Taxonomy

Taxonomic issues: Colonies of *Dichocoenia stokesi* from lower reef slopes or shaded habitats have markedly smaller corallites than those from more exposed environments and are usually identified as *Dichocoenia stellaris* (Wells, 1973). The petition cites the IUCN species account in differentiating these two species; hence, this Status Review Report addresses *Dichocoenia stokesi*.

Family: Meandrinidae.

Evolutionary and geologic history: The genus *Dichocoenia* dates from at least the Oligocene Era in the Caribbean region (Edinger and Risk, 1995).

Global Distribution

Dichocoenia stokesi is restricted to the west Atlantic where it occurs throughout the Caribbean, the Gulf of Mexico, Florida (including the Florida Middle Grounds), the Bahamas, and Bermuda (IUCN Species account). S. dePutron (Bermuda Institute of Ocean Sciences, St. George's. pers. comm., May 2010) confirmed the presence of *Dichocoenia stokesi* in Bermuda and categorized its abundance as rare. T. Murdoch (Bermuda Zoological Society, Flatts. pers. comm.. May 2010) also confirmed its occurrence as rare and added that it is mainly found on forereefs at depths of 10–27 m where he noticed it being partially-to-fully bleached.



Figure 6.4.3. *Dichocoenia stokesi* distribution from IUCN copied from <http://www.iucnredlist.org>



Figure 6.4.4. *Dichocoenia stokesi* distribution from Veron and Stafford-Smith (2002).

U.S. Distribution

Dichocoenia stokesi occurs throughout U.S. waters in the western Atlantic, including the Gulf of Mexico, Florida, Puerto Rico, and the Virgin Islands. Within federally protected U.S. waters, the species has been recorded from the following areas:

- Florida Keys National Marine Sanctuary
- Flower Garden Banks National Marine Sanctuary
- Navassa National Wildlife Refuge
- Dry Tortugas National Park
- Biscayne National Park
- Virgin Islands National Park/Monument
- Buck Island National Monument

Habitat

Habitat: *Dichocoenia stokesi* is found in most reef environments within its range (Veron, 2000), including both backreef and forereef environments, rocky reefs, lagoons, spur-and-groove formations, channels, and occasionally at the

base of reefs (IUCN Species Account). When found in exposed reefs at depths less than 20 m, its hemispherical heads are more abundant than usual (IUCN, 2010).

Depth range: *Dichocoenia stokesi* has been reported in water depths ranging from 2 to 72 m (Carpenter et al., 2008). This considerable depth range suggests the potential for deep refugia, but it is not likely that it attains high abundance at deeper depths.

Abundance

Dichocoenia stokesi is usually uncommon (Veron, 2000). The overall colony density of *Dichocoenia stokesi* averaged across all habitat types in the south Florida region was ~ 1.6 colonies per 10 m², making it the ninth most abundant coral species in this region (Wagner et al., 2010). Substantial population declines have been reported from a bay in Curaçao (80% decline between 1961 and 1992; Debrot et al., 1998) and the upper Florida Keys (see disease description below). There have been no obvious trends in the abundance of *Dichocoenia stokesi* in monitoring of randomized stations at La Parguera, Puerto Rico, St. John, nor St Croix USVI with less than 1.5% cover at most sites (NOAA-Center for Coastal Monitoring and Assessment; http://www8.nos.noaa.gov/biogeopublic/query_habitat.aspx)

Life History

Reproductive characteristics of *Dichocoenia stokesi* have been described from a histological study of populations in southeast Florida (Hoke, 2007). This species is predominantly a gonochoric spawner with an overall sex ratio of 2:1 (male:female), but a small portion of hermaphroditic colonies (~ 18%) were observed in this population. Mean egg size is reported at 312.2 µm (SD 40) and fecundity as 1138 eggs per cm² per year. Minimum colony size at reproduction was found to be 160 cm² in this population and two potential spawning events per annum were inferred: one in late August/early September and a second in October.

Bak and Engel (1979) reported very low densities of *Dichocoenia* juveniles (approximately 1% of total juvenile colonies). However, reports of juveniles of *Dichocoenia stokesi* have been relatively common compared to most other scleractinian corals in the Florida Keys with mean juvenile densities among 566 sites surveyed during 1999–2009 averaging 0.11 per m², but reaching densities as high as 1 juvenile per m² in certain habitats (Chiappone, 2010).

The annual growth rate of *Dichocoenia stokesi* has been reported to increase 2–7 mm per year in diameter and increase 2–5.2 mm per year in height (Vaughn, 1915).

The mounding morphology and large corallite diameter of *Dichocoenia stokesi* enhance turbulence near the surface of colonies (Gardella and Edmunds, 2001). This should, in turn, enhance mass transfer, which affects photosynthesis and respiration in *Dichocoenia stokesi* (Gardella and Edmunds, 1999) as well as prey capture and nutrient uptake. Thresholds for uptake of inorganic nitrogen in *Dichocoenia stokesii* have been reported to be fairly low (150 nM; Davis and Jones, 1997), giving it a potential advantage in nutrient-poor conditions.

Threats

Thermal stress: Although *Dichocoenia stokesi* is susceptible to bleaching (loss of zooxanthellae), it showed the lowest bleaching response (of species observed to bleach) in the south Florida region (Wagner et al., 2010), and in Barbados it ranked 16th of 21 species in bleaching prevalence (Oxenford et al., 2008) during the 2005 Caribbean mass-bleaching event. It was also observed to be bleaching-tolerant in the U.S. Virgin Islands during the same event (Clark et al., 2009). Hence, this species is regarded to be at relatively low threat from temperature-induced bleaching. *Dichocoenia stokesi* hosts clade B zooxanthellae (Correa et al., 2009; LaJeunesse, 2002). Zooxanthellae in clade B do not grow well at high temperatures (Kinzie et al., 2001), but in the field corals with this clade may be relatively bleaching-resistant (McField, 1999). Experimental studies suggest clade B is more bleaching-resistant than clade C, but less resistant than clade A (Warner et al., 2006).

Acidification: No specific research has addressed the effects of acidification on the genus *Dichocoenia*. However, most corals studied have shown negative relationships between acidification and growth (Table 3.2.2), and acidification is likely to contribute to reef destruction in the future (Hoegh-Guldberg et al. 2007, Silverman et al. 2009). While ocean acidification has not been demonstrated to have caused appreciable declines in coral populations so far, the BRT considers it to be a significant threat to corals by 2100 (Albright et al., 2010; Hoegh-Guldberg et al., 2007; Langdon and Atkinson, 2005; Manzello, 2010; Silverman et al., 2009).

Disease: *Dichocoenia stokesi* has been reported to be highly susceptible to white plague (see Fig. 6.4.2), with infection increasing with temperature (Borger and Steiner, 2005). An outbreak event for this disease in the Florida Keys had demonstrable impact at the local population level, yielding mortality of 75% of colonies across several reef sites, substantial shifts in population structure, and essentially no recovery over a 7-year follow-up period (Richardson and Voss, 2005). This species has also been reported to be susceptible to black-band disease (Sutherland et al., 2004), ciliate infection (Croquer et al., 2006), and dark-spot syndrome (Borger and Steiner, 2005). However, disease susceptibility appears to be variable (Borger and Steiner, 2005); for example, *Dichocoenia stokesi* was minimally affected during a 1998 outbreak in St. Lucia that caused widespread mortality in *Montastraea faveolata* and other species (Nugues, 2002).

Predation: *Dichocoenia stokesi* is minimally affected by predation. It can be heavily bioeroded, particularly by bivalves (Highsmith, 1981), and lose substantial amounts of tissue to sponge overgrowth (Hill, 1998).

Land-based sources of pollution (LBSP): One laboratory study has shown that *Dichocoenia stokesi* displays physiological stress at turbidity levels that are within allowable levels as regulated by the State of Florida for coastal construction projects. While light levels and photosynthesis were not affected, respiration levels and mucous production were significantly higher at turbidity levels as low as 14–16 NTU, and P:R fell below 1 at 28–30 NTU (Telesnicki and Goldberg, 1995). An earlier laboratory study examining oil/sediment rejection indicated that *Dichocoenia stokesi* was intermediate (of 19 Caribbean coral species examined) in the rate of sediment removal from its tissues (Bak and Elgershuizen, 1976).

LBSP-related stresses (nutrients, sediment, toxins, and salinity) often act in concert rather than individually and are influenced by other biological (e.g., herbivory) and hydrological factors. Collectively, LBSP stresses are unlikely to produce extinction at a global scale; however, they may pose significant threats at local scales and reduce the resilience of corals to bleaching (Carilli et al., 2009a; Wooldridge, 2009b).

Collection/Trade: Collection and trade are not considered a threat to *Dichocoenia stokesi* (CITES, 2010).

Risk Assessment

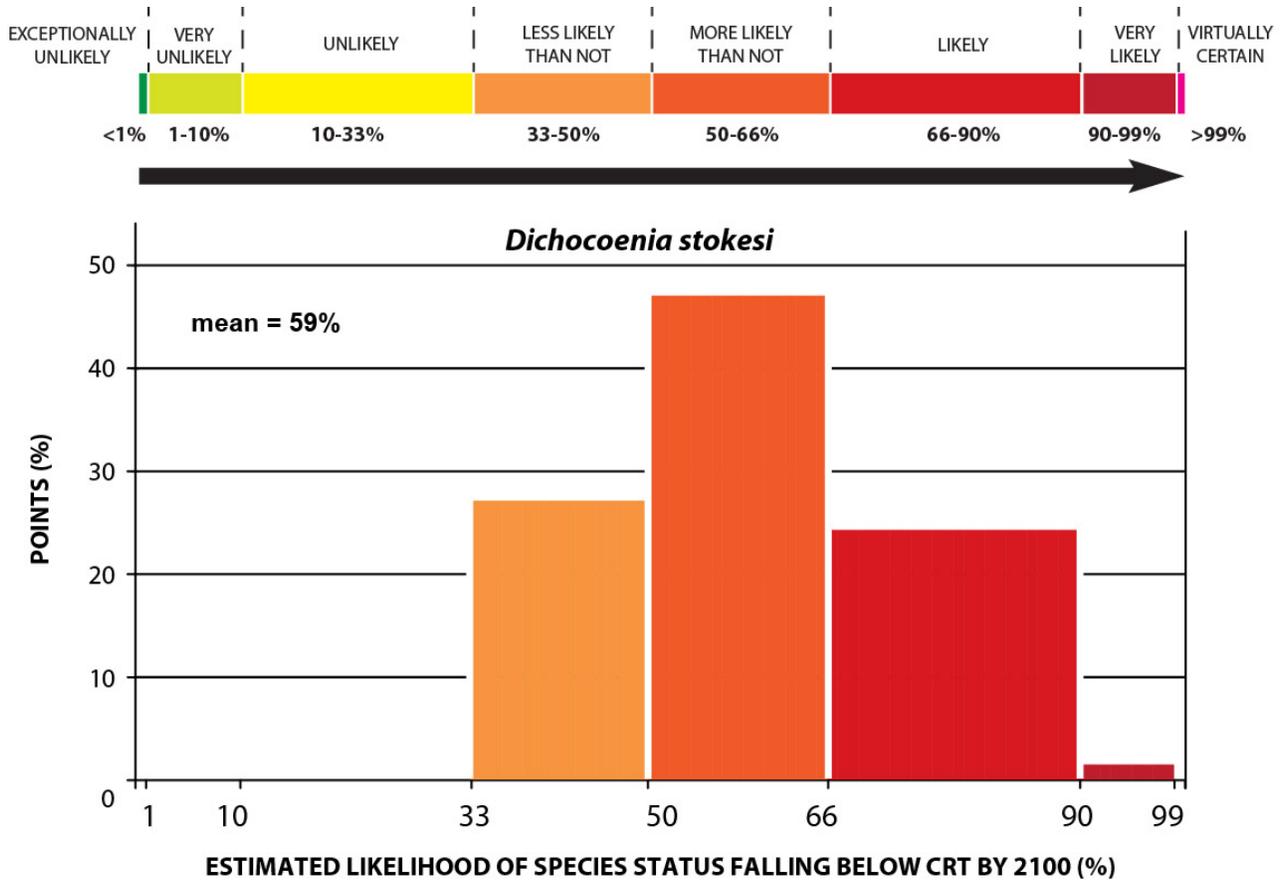


Figure 6.4.5. Distribution of points to estimate the likelihood that the status of *Dichocoenia stokesi* falls below the Critical Risk Threshold (the species is of such low abundance or so spatially fragmented or at such reduced diversity that extinction is extremely likely) by 2100.

Factors that increase the potential extinction risk (higher likelihood of falling below the Critical Risk Threshold) for *Dichocoenia stokesi* include its documented population-level impacts from disease. Factors that reduce potential extinction risk (decrease the likelihood of falling below Critical Risk Threshold) are its relatively high abundance and persistence across many habitat types, including nearshore and mesophotic reefs. Residency in a wide range of habitat types suggests the species has a wide tolerance to environmental conditions and, therefore, better capacity to deal with changing environmental regimes.

The overall likelihood that *Dichocoenia stokesi* will fall below the Critical Risk Threshold by 2100 was estimated to be in the “more likely than not” risk category with a mean likelihood of 59% and a standard error (SE) of 5.1% (Fig. 6.4.5). This SE was calculated by taking the standard deviation of the seven mean voting scores of the BRT members and shows the highest degree of coherence among the BRT for any of the 82 candidate species. The uncertainty of the BRT is reflected in the range of votes of 33%–99% (Fig. 6.4.5) and the relatively high average range of likelihood estimates of the seven BRT voters (58.3%). The overall moderate extinction risk estimate placed it at lower risk than most other western Atlantic candidates.

6.5 Genus *Montastraea* (Family Faviidae)

Montastraea annularis complex

Taxonomic Issues

The nominal *Montastraea annularis* (Ellis and Solander, 1786) has historically been one of the primary reef framework builders of the western Atlantic and Caribbean. *Montastraea annularis* and its related species seem to have originated prior to the Caribbean coral extinction at the end of late Pliocene to early Pleistocene (~ 2.5 Ma; Budd and Klaus, 2001). Its depth range is from 1 m to over 30 m, and has historically been considered a highly plastic species with multiple growth forms ranging from columnar, to massive, to platy. In the early 1990s, Knowlton, Weil, and colleagues suggested the partitioning of these growth forms into separate species, resurrecting previously described monikers, *Montastraea faveolata* and *Montastraea franksi*. These three sibling species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowlton, 1994). Subsequent reproductive and genetic studies have generally supported this partitioning, although with some interesting details. *Montastraea faveolata* is the most genetically distinct, while *Montastraea annularis* and *Montastraea franksi* are less so (Fukami et al., 2004; Lopez et al., 1999). Similarly, hybrid-crossing experiments show the same pattern with *Montastraea annularis* and *Montastraea franksi* showing greater success of hybrid crosses than either with *Montastraea faveolata* (Leviton et al., 2004; Szmant et al., 1997). Isolation between *Montastraea annularis* and *Montastraea franksi* is enhanced by the timing of spawning; *Montastraea franksi* spawns 1–2 hours earlier than the other two. Meanwhile, Fukami et al. (2004) showed some degree of geographic variation in these reproductive and genetic traits with a lesser degree of separation in the Bahamas than in Panama.

While there now is reasonable acceptance that these represent three valid species, long-term monitoring data sets and previous ecological studies did not distinguish among them. Currently, intermediate forms (especially in northern sections of the range) complicate the collection of monitoring data into three species, and so modern monitoring data sets often still group them as “*Montastraea annularis* complex” or “*Montastraea annularis sensu lato*.” The BRT has estimated Critical Risk Thresholds separately for each species, but much of the information available is for the complex as a whole.

Abundance and Trends

The *Montastraea annularis* complex has historically been a dominant species on Caribbean coral reefs, characterizing the so-called “buttress zone” and “annularis zone” in the classical descriptions of Caribbean reefs (Goreau, 1959). Goreau describes *Montastraea annularis* complex as “very abundant” in these zones and constitutes “by far the commonest and often the only fossil framework coral to be found in exposures of the Pliocene Era and more recent coastal reef limestones of northern Jamaica” (Goreau, 1959). There is ample evidence that it has declined dramatically throughout its range, but perhaps at a slower pace than its fast-paced Caribbean colleagues, *Acropora palmata* and *Acropora cervicornis*. While the latter began their rapid declines in the early-to-mid-1980s, declines in *Montastraea annularis* complex have been much more obvious in the 1990s and 2000s, most often associated with combined disease and bleaching events. It should be noted that, given the dramatically low productivity of the *Montastraea annularis* complex (low growth and extremely low recruitment), any substantial declines in adult populations would suggest increased extinction risk since their capacity for population recovery is extremely limited. Figure 6.5 shows only recent trends in aspects of the *Montastraea annularis* complex abundance at select locations, and additional supporting information on longer-term trends is described below. In most cases where examined, additional demographic changes accompany these instances of declining abundance (e.g., size structure of colonies, partial mortality, etc).

In Florida, the percent cover data from four fixed sites have shown the *Montastraea annularis* complex to have declined in absolute cover from 5% to 2% in the Lower Keys between 1998 and 2003 (Fig. 6.5A) and was accompanied by 5–40% colony shrinkage and virtually no recruitment (Smith et al., 2008). Earlier studies from the Florida Keys indicated a 31% decline of *Montastraea annularis* complex absolute cover between 1975 and 1982 (Dustan and Halas, 1987) at Carysfort Reef and > 75% decline (from over 6% cover to less than 1%) across several sites in Biscayne National Park between the late 1970s and 1998–2000 (Dupont et al., 2008). Taken together, these data imply extreme declines in the Florida Keys (80%–95%) between the late 1970s and 2003, and it is clear that further dramatic losses occurred in this region during the cold weather event in January 2010.

Similar declines have also been documented for relatively remote Caribbean reefs. At Navassa Island National Wildlife Refuge, percent cover of *Montastraea annularis* complex on randomly sampled patch reefs declined from 26% in 2002 to 3% in 2009 (Fig. 6.5B), following disease and bleaching events in this uninhabited oceanic island (Miller and Williams, 2007). Additionally, two offshore islands west of Puerto Rico (Mona and Desecheo; Fig. 6.5C) showed reductions in live colony counts of 24% and 32% between 1998/2000 and 2008 (Bruckner and Hill, 2009). At Desecheo, this demographic decline of one-third corresponded to a decline in *Montastraea annularis* complex cover from over 35% to below 5% across 4 sites. Taken together, decadal-scale declines across these remote islands in the central Caribbean constitute over 85% of the populations.

In the U.S. Virgin Islands, recent data from the U.S. National Park Service Inventory and Monitoring Program (Fig. 6.5D) across six sites at fixed stations show a decline of *Montastraea annularis* complex from just over 10% cover in 2003 to just over 3% cover in 2009 following mass bleaching and disease impacts in 2005 (Miller et al., 2009). This degree of recent decline was preceded by a decline from over 30% *Montastraea annularis* complex cover to ~ 10% between 1988 and 2003 as documented by Edmunds and Elahi (2007). Similarly, percent cover of *Montastraea annularis* complex in a marine protected area in Puerto Rico declined from 49% to 8% between 1997 and 2009 (Hernández-Pacheco et al., 2011). Taken together, these data suggest an 80%–90% decline in *Montastraea annularis* complex over the past two decades in the main U.S. Caribbean territories.

While Bak and Luckhurst (1980) indicated stability in *Montastraea annularis* complex cover across depths in Curaçao during a 5-year study in the mid-1970s, this region has also manifested *Montastraea annularis* complex declines in recent years. Bruckner and Bruckner (2006) documented an 85% increase in the partial mortality of *Montastraea annularis* complex colonies across three reefs in western Curaçao between 1998 and 2005 (Fig. 6.5E), approximately twice the level for all other scleractinian species combined. These authors noted that *Montastraea franksi* fared substantially better than the other two complex species in this study. It is likely that *Montastraea annularis* complex populations in Curaçao have fared better than other Caribbean regions but are not immune to losses.

Montastraea annularis complex declines in additional locations can be noted. For example, at Glovers Reef, Belize (McClanahan and Muthiga, 1998) documented a 38%–75% decline in relative cover of *Montastraea annularis* complex across different reef zones between 1975 and 1998, and a further 40% decline in relative cover has occurred since then (Huntington et al., in review). In contrast, *Montastraea annularis* complex populations have shown stable status at sites in Columbia between 1998 and 2003 (Rodriguez-Ramirez et al., 2010), although demographic changes in *Montastraea annularis* complex at both degraded and less-degraded reefs imply some degree of population decline in this region (Alvarado-Chacon and Acosta, 2009).

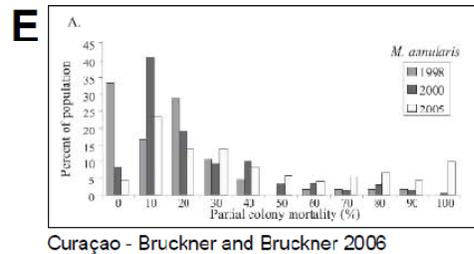
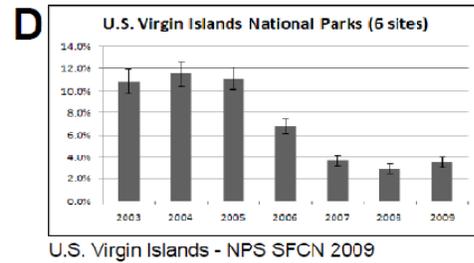
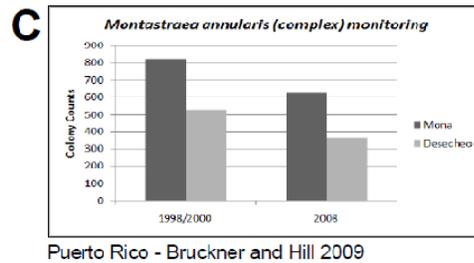
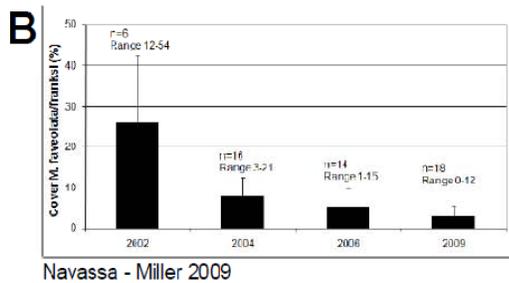
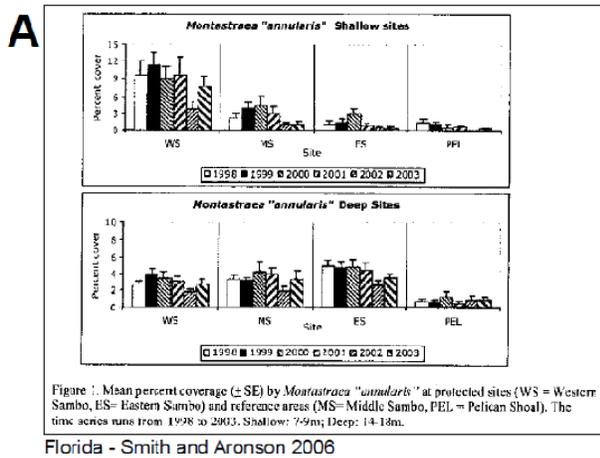
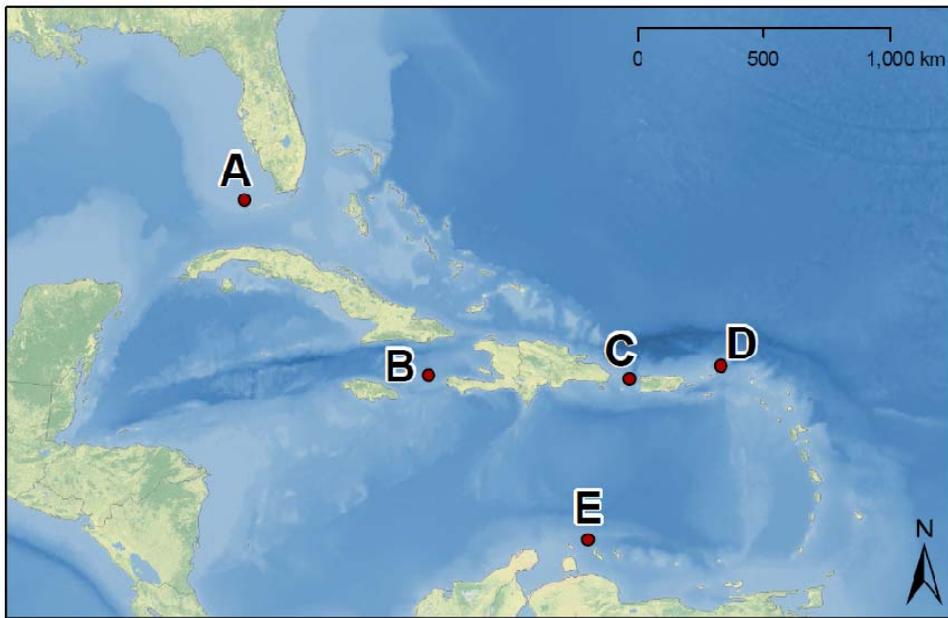


Figure 6.5. Examples of declining abundance of *Montastraea annularis* complex in different regions of the Caribbean in the recent past. A) data from Smith and Aronson (Smith et al., 2006) based on haphazard video transects at two depths at 4 sites in the lower Florida Keys; B) unpublished data from haphazard photo quadrats at randomly selected patch reefs at Navassa Island National Wildlife Refuge (described in (Miller et al., 2005)); C) declines in colony abundance in fixed plots at two offshore islands in Puerto Rico between 1998/2000 and 2008 (Bruckner and Hill, 2009); D) cover data from video transects of six fixed sites in Virgin Islands National Park (unpublished data, National Park Service, South Florida/Caribbean Network); E) increasing proportion of population across three sites in western Curaçao manifesting high levels of partial mortality (Bruckner and Bruckner, 2006) which is accompanied by a lack of recruitment.

Life History

All three of the *Montastraea annularis* complex species are hermaphroditic broadcast spawners, with spawning concentrated on nights 6–8 following the new moon in late summer (Levitan et al., 2004). Fertilization success measured in the field was generally below 15% but was highly linked to the number of colonies observed spawning at the same time (Levitan et al., 2004). Minimum size for reproduction was found to be 83 cm² in Puerto Rico whether as an intact adult or as a remnant fragment of an older colony (Szmant-Froelich, 1985). Szmant-Froelich (1985) estimated this to correspond to 4–5 years of age, and *Montastraea annularis* typically exhibit a linear growth of ~ 1 cm per year (Gladfelter et al., 1978), but increased appreciation for the slow rate of growth of post-settlement stages suggest this age for minimum reproductive size may be an underestimate (M.W. Miller, Southeast Fisheries Science Center, Miami, FL. pers. obs., October 2010). Eggs (~ 310 to 340 µm among the three species; Szmant et al., 1997) and larvae are small and post-settlement growth rates are very slow, both of which may contribute to extremely low post-settlement survivorship, even lower than other Caribbean broadcasters, such as the threatened Caribbean *Acropora palmata* (Szmant and Miller, 2005). There may be a depth-related fecundity cost arising from morphological differences in polyp spacing (Villinski, 2003), suggesting the spatial distribution of colonies may influence population fecundity on a reef.

Successful recruitment by *Montastraea annularis* complex species have seemingly always been rare events. Hughes and Tanner reported the occurrence of only a single recruit for these species over 18 years of intensive observation of 12 m² of reef in Discovery Bay, Jamaica (Hughes and Tanner, 2000) while myriad other recruitment studies from throughout the Caribbean also report them to be negligible to absent (Bak and Engel, 1979; Rogers et al., 1984). Edmunds (2011) asserted that the large, rare, replenishing recruitment hypothesized to operate in these species have never actually been documented on any Caribbean reef since the initiation of quantitative ecological study in the 1960s. Overall recruitment by these species is so low that Edmunds (2011) based an entire publication on the detection of at most nine additional juvenile colonies (constituting a “recruitment pulse”) along the south shore of St. John, USVI in 2008–2009. However, this “recruitment pulse” was limited in spatial extent (Edmunds et al., 2011). *Montastraea* juveniles also have higher mortality rates than larger colonies (Smith et al., 2006). Despite their generally massive form, at least the lobate form (*Montastraea annularis* sensu stricto) is capable of some degree of fragmentation/fission and clonal reproduction. Foster et al. (2007) detected 8% of *Montastraea annularis* genotypes were represented by multiple ramets (up to 14 ramets or separate colonies of the same genotypes) across three sites in Belize.

In St. Croix, growth rates of *Montastraea annularis* were measured along a depth gradient from 3 m to 40 m (Hubbard and Scaturo, 1985). There was a sharp decline in growth rate at a depth of around 15 m with growth rates of 0.7–0.9 cm per year in water depths < 12 m and 0.20 cm per year in depths below 18–20 m. Growth rates, measured as extension rates, in shallow waters (< 15 m) varied between 0.43 and 1.23 cm per year and in deeper waters (> 18 m) between 0.06 and 0.29 cm per year. Also, growth rates were consistently higher in the clear waters of Cane Bay than those at the more turbid and sediment rich waters of Salt River confirming the controlling factors for growth rate of light and sediment load (Hubbard and Scaturo, 1985). Long-term analyses of coral cores have typically shown seasonal variation in growth and a general reduction in *Montastraea* growth rates over the past century, although the reduction may have stabilized over the past few decades (Carricart-Ganivet et al., 2000; Dodge and Lang, 1983; Hudson et al., 1994).

Threats

Because they have traditionally been common and are one of the main reef builders in the Caribbean, *Montastraea annularis* complex species have been the frequent subject of research attention, including responses to and impacts of environmental threats. This body of work is briefly summarized here, but it should be noted that a large body of work exists for these species.

Thermal stress: Published reports of individual bleaching surveys have consistently indicated that *Montastraea annularis* complex is highly-to-moderately susceptible to bleaching (Brandt, 2009; Bruckner and Hill, 2009; Oxenford et al., 2008; Wagner et al., 2010). The species complex is polymorphic with respect to zooxanthellae. Depending on depth and other environmental conditions, colonies can contain clade A, B, C, D, but composition of symbiont assemblages in at least some areas changes in response to bleaching (Rodríguez-Román et al., 2006; Thornhill et al., 2006). Bleaching has been shown to prevent gametogenesis in *Montastraea annularis* complex colonies in the following reproductive season after recovering normal pigmentation (Mendes and Woodley, 2002; Szmant and Gassman, 1990) and leave permanent records in coral growth records (Leder et al., 1991; Mendes and Woodley, 2002). Given the rapidly developing genomic tools for this species complex, cellular and transcriptomic mechanisms for bleaching and thermal stress are being elucidated for this species complex (Desalvo et al., 2008). In addition, certain aspects of geographic and

genetic variability in the molecular responses to thermal stress have been described (Polato et al., 2010), which may enable more accurate predictions of potential evolutionary adaptation to warming. Particularly well-documented mortalities in these species following severe mass-bleaching in 2005 highlight the immense impact that thermal stress events and their aftermath can have on *Montastraea annularis* complex populations (Miller et al., 2009). Using demographic data collected in Puerto Rico over 9 years straddling the 2005 bleaching event (Hernández-Pacheco et al., 2011) showed that demographic transitions (vital rates) for *Montastraea annularis* were substantially altered by the 2005 mass thermal bleaching event. Size-based transition matrix models based on these measured vital rates showed that population growth rates were stable (λ not significantly different from 1) in the pre-bleaching period (2001–2005) but declined to $\lambda = 0.806$ one year after and to 0.747 two years after the bleaching event. Although population growth rate returned to $\lambda = 1$ the following year, simulation modeling of different bleaching probabilities predicted extinction of a population with these dynamics within 100 years at a bleaching probability between 10 and 20%, i.e., once every 5 to 10 years (Hernández-Pacheco et al., 2011). Cervino (2004) also showed that higher temperatures (over experimental treatments from 20°C–31°C) resulted in faster rates of tissue loss and higher mortality in yellow-band affected *Montastraea annularis* complex. Recent work in the Mesoamerican reef system indicated that *Montastraea faveolata* had reduced thermal tolerances in locations and over time (Carilli et al., 2010) with increasing human populations, implying increasing local threats (Carilli et al., 2009a).

Acidification: The only study conducted regarding the impact of acidification on this genus is a field study (Helmle et al., 2011) that did not find any change in *Montastraea faveolata* calcification in field-sampled colonies from the Florida Keys up through 1996. However, most corals studied have shown negative relationships between acidification and growth (Table 3.2.2), and acidification is likely to contribute to reef destruction in the future (Hoegh-Guldberg et al. 2007, Silverman et al. 2009). While ocean acidification has not been demonstrated to have caused appreciable declines in coral populations so far, the BRT considers it to be a significant threat to corals by 2100 (Albright et al., 2010; Hoegh-Guldberg et al., 2007; Langdon and Atkinson, 2005; Manzello, 2010; Silverman et al., 2009). Preliminary experiments testing effects of acidification on fertilization and settlement success of *Montastraea annularis* complex (Albright et al., unpublished data) show results that are consistent with the significant impairments demonstrated for *Acropora palmata* (Albright et al., 2010).

Disease: Both Bruckner and Hill (2009) and Miller et al. (2009) demonstrated profound population declines for *Montastraea annularis* complex from disease impacts, both with and without prior bleaching. Both white-plague and so-called yellow-band diseases can invoke this type of population level decline. Disease outbreaks can persist for years in a population—*Montastraea annularis* colonies suffering from yellow-band in Puerto Rico in 1999 still manifest similar disease signs 4 years later, with a mean tissue loss of 60% (Bruckner and Bruckner, 2006).

Predation: *Montastraea annularis* complex does not suffer from catastrophic outbreaks of predators, such as the effects of *Acanthaster planci* on *Acropora* stands in the Pacific. While *Montastraea annularis* complex can host large populations of corallivorous snails, they rarely display large feeding scars that are apparent on other coral prey, possibly related to differences in tissue characteristics or nutritional value (Baums et al., 2003). However, low-level predation can have interactive effects with other stressors. For example, predation by butterflyfish can serve as a vector to facilitate infection of *Montastraea faveolata* with black-band disease (Aeby and Santavy, 2006). Parrotfishes are also known to preferentially target *Montastraea annularis* complex in so-called “spot-biting” which can leave dramatic signs in some local areas (Bruckner et al., 2000; Rotjan and Lewis, 2006), and chronic parrotfish biting can impede colony recovery from bleaching (Rotjan et al., 2006).

Although it is not predation per se, *Montastraea* colonies have often been infested by other pest organisms. Bioeroding sponges (Ward and Risk, 1977) and territorial damselfishes, *Stegastes planifrons*, can cause tissue loss and skeletal damage. Damselfish infestation of *Montastraea annularis* complex appears to have increased in areas where their preferred, branching coral habitat has declined because of loss of Caribbean *Acropora* spp. (Precht et al., 2010).

Land-based sources of pollution (LBSP): Large, massive, long-lived colonies of *Montastraea annularis* complex lend themselves to retrospective studies of coral growth in different environments so there is a relatively large amount known or inferred regarding relationships of water quality to *Montastraea annularis* complex growth and status. For example, Tomascik (1990) found an increasing average growth (linear extension) rate of *Montastraea annularis* with improving environmental conditions on fringing reefs in Barbados. Tomascik also found a general pattern of decreasing growth rates within the past 30 years at each of the 7 fringing reefs and contributed this decrease to the deterioration of water quality along the west coast of Barbados. Torres and Morelock (2002) noted a similar decline in *Montastraea annularis*

growth at sediment-impacted reefs in Puerto Rico. Density and calcification rate increased from high to low turbidity and sediment load, while extension rate followed an inverse trend (Carricart-Ganivet and Merino, 2001). Eakin et al. (1994) demonstrated declines in *Montastraea annularis* linear extension during periods of construction in Aruba.

Downs et al. (2005) suggested that localized toxicant exposure may account for a localized mortality event of *Montastraea annularis* complex in Biscayne National Park, based on analyses of a suite of cellular biomarkers that yielded signatures of oxidative stress and xenobiotic detoxification response. Meanwhile, *Montastraea annularis* complex was shown to have somewhat lesser sensitivity to copper exposure in laboratory assays than *Acropora cervicornis* and *Pocillopora damicornis* (Bielmyer et al., 2010). *Montastraea faveolata* induces cytochrome p450 and antioxidant enzymes under acute exposure to benzo(a)pyrene (Ramos and Garcia, 2007), but effects of chronic long-term exposure are not known. *Montastraea annularis* skeletons are among those that incorporate toxic heavy metals, making them useful in documenting long-term contamination of reef sites (Medina-Elizalde et al., 2002; Runnalls and Coleman, 2003).

Nutrient-related runoff has also been deleterious to *Montastraea annularis* complex. Elevated nitrogen reduced respiration and calcification in *Montastraea annularis* and stimulated zooxanthellae populations (Marubini and Davies, 1996). Fecal coliform microorganisms were among the bacterial communities associated with *Montastraea* in the Florida Keys (Lipp et al., 2002), suggesting potential sewage impacts to the corals. Elevated nutrients increased the rate of tissue loss in *Montastraea franksi* and *Montastraea faveolata* affected by yellow-band disease (Bruno et al., 2003). Chronic nutrient elevation can produce bleaching and partial mortality in *Montastraea annularis*, whereas anthropogenic dissolved organic carbon kills corals directly (Kuntz et al., 2005).

Overall, LBSP-related stresses (nutrients, sediment, toxins, and salinity) often act in concert rather than individually and are influenced by other biological (e.g., herbivory) and hydrological factors. Collectively, LBSP stresses are unlikely to produce extinction at a global scale; however, they may pose significant threats at local scales and reduce the resilience of corals to bleaching (Carilli et al., 2009a; Wooldridge, 2009b).

Collection/Trade: *Montastraea annularis* complex species have a very low occurrence in the CITES trade databases (CITES, 2010). Hence, collection/trade is not considered to be a significant threat to *Montastraea annularis* complex species.

6.5.1 *Montastraea faveolata* Ellis and Solander, 1786



Figure 6.5.1. *Montastraea faveolata* photo (left) from Veron and Stafford-Smith (2002) and (right) polyp view. Photo from the NOAA Southeast Fisheries Science Center.

Characteristics

Montastraea faveolata grows in heads or sheets, the surface of which may be smooth or have keels or bumps. Septa are highly exsert and the skeleton is much less dense than in the other two *Montastraea* species (Weil and Knowton, 1994). Colony diameter can reach up to 10 m with a height of 4–5 m (Szmant et al., 1997). Common colors are grey, green, and brownish (Szmant et al., 1997).

Taxonomy

Taxonomic issues: See Section 6.5: “Genus *Montastraea*.” Veron (2000) does not list *Montastraea faveolata* separately from the *Montastraea annularis* complex.

Family: Faviidae.

Evolutionary and geologic history: The *Montastraea* genus arose in the Oligocene ~ 30 Ma (Edinger and Risk, 1995).

Global Distribution

The range of *Montastraea faveolata* is restricted to the west Atlantic. According to both the IUCN Species Account and the CITES species database, *Montastraea faveolata* occurs throughout the Caribbean, including Bahamas, Flower Garden Banks and the entire Caribbean coastline, but there are no records from Bermuda. S. dePutron (Bermuda Institute of Ocean Sciences, St. George’s. pers. comm., May 2010) confirmed the presence of *Montastraea faveolata* in Bermuda and categorized its abundance as common. T. Murdoch (Bermuda Zoological Society, Flatts. pers. comm., May 2010) also confirmed its occurrence but listed it as rare and added that it has probably suffered a substantial loss from the 1995 yellow-band outbreak.

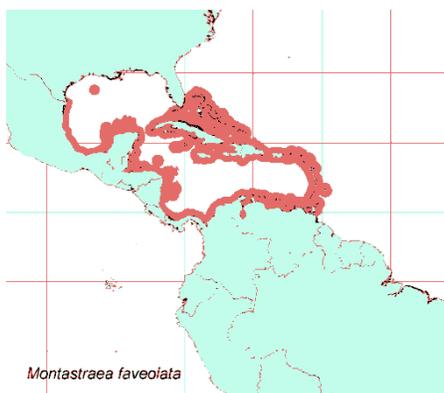


Figure 6.5.2. *Montastraea faveolata* distribution from IUCN copied from <http://www.iucnredlist.org>.

U.S. Distribution

Montastraea faveolata is common throughout the U.S. waters of the west Atlantic and greater Caribbean region and is present within federally protected waters, including:

- Flower Garden Banks National Marine Sanctuary
- Florida Keys National Marine Sanctuary
- Dry Tortugas National Park
- Virgin Island National Park/Monument
- Biscayne National Park
- Navassa Island National Wildlife Refuge
- Buck Island Reef National Monument

Habitat

Habitat: *Montastraea faveolata* has been reported in most reef habitats, often the most abundant coral between 10 and 20 m in forereef environments.

Depth range: *Montastraea faveolata* has been reported in water depths ranging from 0.5 m to 40 m (Carpenter et al., 2008; Weil and Knowton, 1994). *Montastraea* spp. are a common, often dominant component of Caribbean mesophotic reefs (Smith et al., 2010), suggesting the potential for deep refugia.

Abundance

See Section 6.5: “Genus *Montastraea*.”

Life History

See Section 6.5: “Genus *Montastraea*.”

In many life history characteristics, including growth rates, tissue regeneration, egg size, *Montastraea faveolata* is considered to be intermediate between its two sister species (Szmant et al., 1997).

Threats

See Section 6.5: “Genus *Montastraea*.”

Risk Assessment

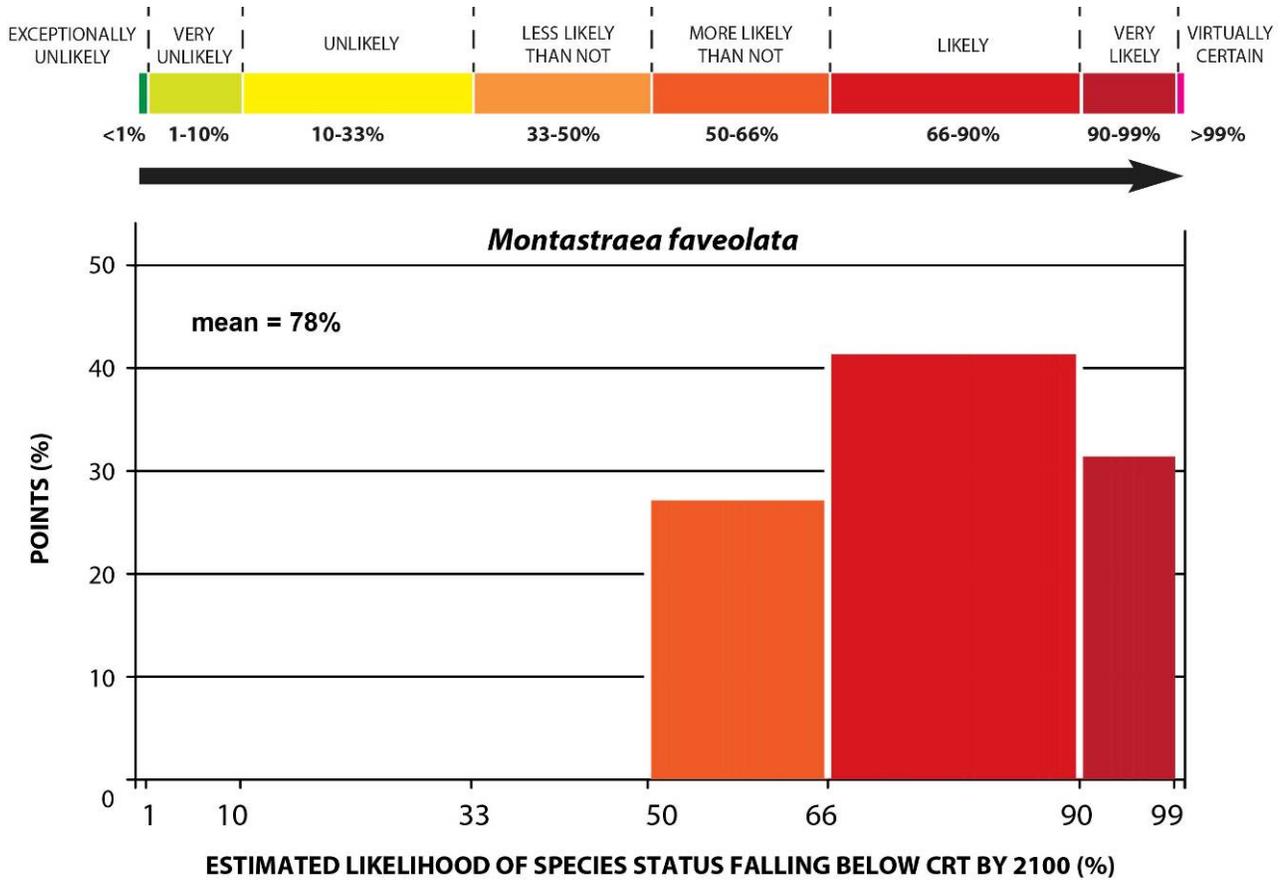


Figure 6.5.3. Distribution of points to estimate the likelihood that the status of *Montastraea faveolata* falls below the Critical Risk Threshold (the species is of such low abundance or so spatially fragmented or at such reduced diversity that extinction is extremely likely) by 2100.

Factors that increase the potential extinction risk (higher likelihood of falling below the Critical Risk Threshold) for *Montastraea faveolata* were its extremely low productivity (growth and recruitment) documented dramatic recent declines and its restriction to the highly disturbed/degraded wider Caribbean region. All these factors combine to yield a very high estimated extinction risk.

The overall likelihood that *Montastraea faveolata* will fall below the Critical Risk Threshold by 2100 was estimated to be in the “likely” risk category with a mean likelihood of 78% and a standard error (SE) of 7% (Fig. 6.5.3). This SE was calculated by taking the standard deviation of the seven mean voting scores of the BRT members and shows the coherence among the BRT. The uncertainty of the BRT is reflected in the range of votes of 50%–99% (Fig. 6.5.3) and the average range of likelihood estimates of the seven BRT voters (45%). This overall range of votes (spanning only three risk categories) was small relative to most of the other candidate species.

6.5.2 *Montastraea franksi* Gregory, 1895



Figure 6.5.4. *Montastraea franksi* photo (left) from Veron and Stafford-Smith (2002) and (right) from http://sanctuaries.noaa.gov/pgallery/pgflower/living/living_2.html.

Characteristics

Montastraea franksi is distinguished by large, unevenly-arrayed polyps that give the colony its characteristic irregular surface. Colony form is variable, and the skeleton is dense with poorly developed annual bands (Weil and Knowton, 1994). Colony diameter can reach up to 5 m with a height of up to 2 m (Szmant et al., 1997). Common colors are green, grey, and brown (Szmant et al., 1997).

Taxonomy

Taxonomic issues: See Section 6.5: “Genus *Montastraea*.” Veron (2000) does not list *Montastraea franksi* separately from the *Montastraea annularis* complex.

Family: Faviidae.

Evolutionary and geologic history: The *Montastraea* genus arose in the Oligocene ~ 30 Ma (Edinger and Risk, 1995).

Global Distribution

Montastraea franksi is found throughout the Caribbean Sea, including in the Bahamas, Bermuda, and Flower Garden Banks. The range is restricted to the west Atlantic and there is no range fragmentation.

According to both the IUCN Species Account and the CITES species database, *Montastraea franksi* occurs throughout the Caribbean, including the Bahamas, Flower Garden Banks, Bermuda, and the entire Caribbean coastline. S. dePutron (Bermuda Institute of Ocean Sciences, St. George’s. pers. comm., May 2010) confirmed the presence of *Montastraea franksi* in Bermuda and categorized its abundance as dominant. T. Murdoch (Bermuda Zoological Society, Flatts. pers. comm., May 2010) also confirmed its occurrence but listed it as common and added that it has definitely suffered a substantial loss from the 1995 yellow-band outbreak.

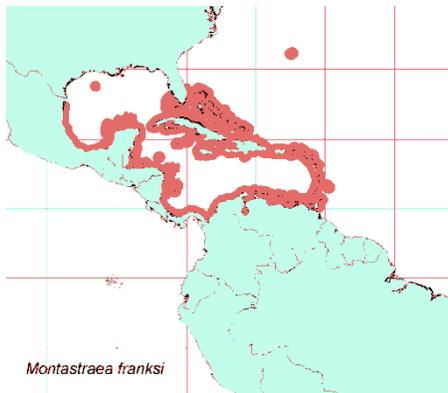


Figure 6.5.5. *Montastraea franksi* distribution from IUCN copied from <http://www.iucnredlist.org>.

U.S. Distribution

Montastraea franksi is widely distributed throughout U.S. waters of the west Atlantic and greater Caribbean, including Florida and the Flower Garden Banks.

Within federally protected waters, *Montastraea franksi* has been recorded from the following areas:

- Flower Garden Banks National Marine Sanctuary
- Florida Keys National Marine Sanctuary
- Dry Tortugas National Park
- Virgin Island National Park/Monument
- Biscayne National Park
- Navassa Island National Wildlife Refuge
- Buck Island Reef National Monument

Habitat

Habitat: *Montastraea franksi* occupies most reef environments (Carpenter et al., 2008).

Depth range: *Montastraea franksi* has been reported from water depths ranging from 5 m to 50 m (Bongaerts et al., 2010; Carpenter et al., 2008; Weil and Knowton, 1994). *Montastraea* spp. are a common, often dominant component of Caribbean mesophotic reefs (Smith et al., 2010), suggesting the potential for deep refugia. *Montastraea franksi* tends to have a deeper distribution than the other two species in the *Montastraea annularis* complex (Szmant et al., 1997).

Abundance

Montastraea franksi has been reported as common (Veron, 2000). See Section 6.5: “Genus *Montastraea*.”

Life History

See Section 6.5: “Genus *Montastraea*.”

The growth rate for *Montastraea franksi* is reported to be slower, and spawning is reported to be about 1 hour earlier than for *Montastraea annularis* and *Montastraea faveolata* (Szmant et al., 1997).

Threats

See Section 6.5: “Genus *Montastraea*.”

Risk Assessment

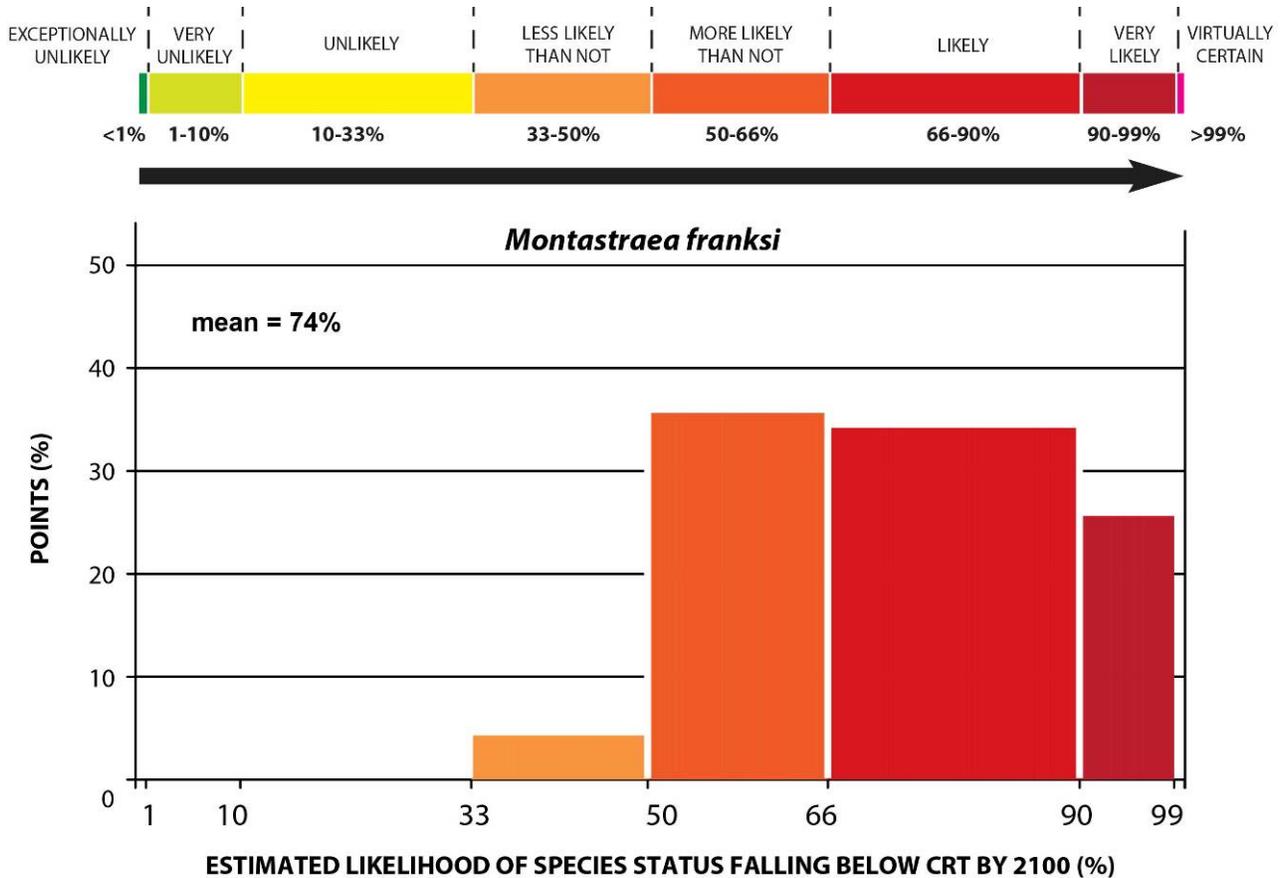


Figure 6.5.6. Distribution of points to estimate the likelihood that the status of *Montastraea franksi* falls below the Critical Risk Threshold (the species is of such low abundance or so spatially fragmented or at such reduced diversity that extinction is extremely likely) by 2100.

Factors that increase the potential extinction risk (higher likelihood of falling below the Critical Risk Threshold) for *Montastraea franksi* were its extremely low productivity (growth and recruitment), documented dramatic recent declines, and its restriction to the highly disturbed/degraded wider Caribbean region. All of these factors combine to yield a very high estimated extinction risk. It had a marginally lower risk estimate than the other two *Montastraea annularis* complex species because of its greater distribution in deep and mesophotic depth habitats, which are expected to experience lesser exposure to some surface-based threats.

The overall likelihood that *Montastraea franksi* will fall below the Critical Risk Threshold by 2100 was estimated to be in the “likely” risk category with a mean likelihood of 74% and a standard error (SE) of 9% (Fig. 6.5.6). This SE was calculated by taking the standard deviation of the seven mean voting scores of the BRT members and shows the coherence among the BRT. The uncertainty of the BRT is reflected in the range of votes of 33%–99% (Fig. 6.5.6) and the average range of likelihood estimates of the seven BRT voters (48%).

6.5.3 *Montastraea annularis* Ellis and Solander, 1786

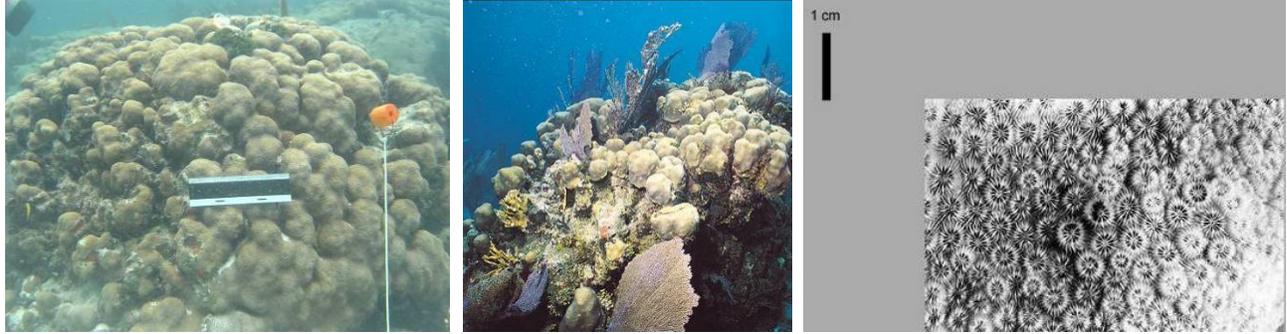


Figure 6.5.7. *Montastraea annularis sensu stricto* photo (middle) and corallite plan from Veron and Stafford-Smith (2002). Large colony photo (left) from NOAA Southeast Fisheries Science Center.

Characteristics

Montastraea annularis colonies grow in columns that exhibit rapid and regular upward growth. In contrast to the other species, margins on the sides of columns are typically senescent (Weil and Knowton, 1994). Live colony surfaces usually lack ridges or bumps. Corallites on tops of columns are closely packed, uniformly distributed, and evenly exsert, with maximum diameters of mature corallites typically 2.1–2.6 mm.

Taxonomy

Taxonomic issues: See Section 6.5: “Genus *Montastraea*.”

Family: Faviidae.

Evolutionary and geologic history: The *Montastraea* genus arose in the Oligocene ~ 30 Ma (Edinger and Risk, 1995).

Global Distribution

Montastraea annularis has a range restricted to the west Atlantic. It can be found throughout the Caribbean, Bahamas, and Flower Garden Banks (Veron, 2000, IUCN), but may be absent from Bermuda (Weil and Knowton, 1994). S. dePutron (Bermuda Institute of Ocean Sciences, St. George’s. pers. comm., May 2010) confirmed the presence of *Montastraea annularis* in Bermuda and categorized its abundance as rare; T. Murdoch (Bermuda Zoological Society, Flatts, pers. comm., May 2010) had not seen this species in Bermuda.

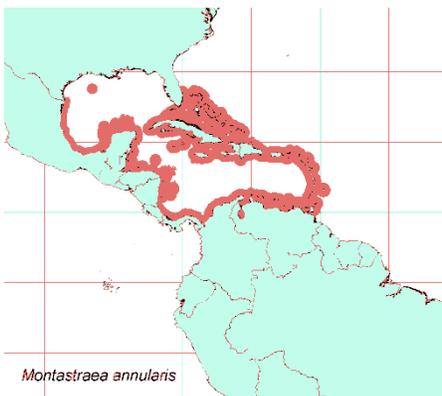


Figure 6.5.8. *Montastraea annularis* distribution from IUCN copied from <http://www.iucnredlist.org>.



Figure 6.5.9. *Montastraea annularis* distribution from Veron and Stafford-Smith (2002).

U.S. Distribution

Montastraea annularis is common throughout U.S. waters of the west Atlantic and greater Caribbean, including Florida and the Gulf of Mexico, within its range including federally protected waters in the following areas:

- Flower Garden Bank Sanctuary
- Dry Tortugas National Park
- Virgin Island National Park/Monument
- Biscayne National Park
- Florida Keys National Marine Sanctuary
- Navassa National Wildlife Refuge
- Buck Island Reef National Monument

Habitat

Habitat: *Montastraea annularis* is reported from most reef environments (Veron, 2000).

Depth range: *Montastraea annularis* has been reported in water depths ranging from 0.5 m to 20 m (Szmant et al., 1997); while *Montastraea* spp. are a common, often dominant component of Caribbean mesophotic reefs (Smith et al., 2010), suggesting the potential for deep refugia. However, *Montastraea annularis* sensu stricto is generally described with a shallower distribution (Szmant et al., 1997).

Abundance

Montastraea annularis has been reported to be common (Veron 2000). See Section 6.5: “Genus *Montastraea*.”

Life History

See Section 6.5: “Genus *Montastraea*.” *Montastraea annularis* is reported to have slightly smaller egg size and potentially smaller size/age at first reproduction than the other two members of the *Montastraea annularis* complex (reviewed in Szmant et al., 1997).

Threats

See Section 6.5: “Genus *Montastraea*.”

Risk Assessment

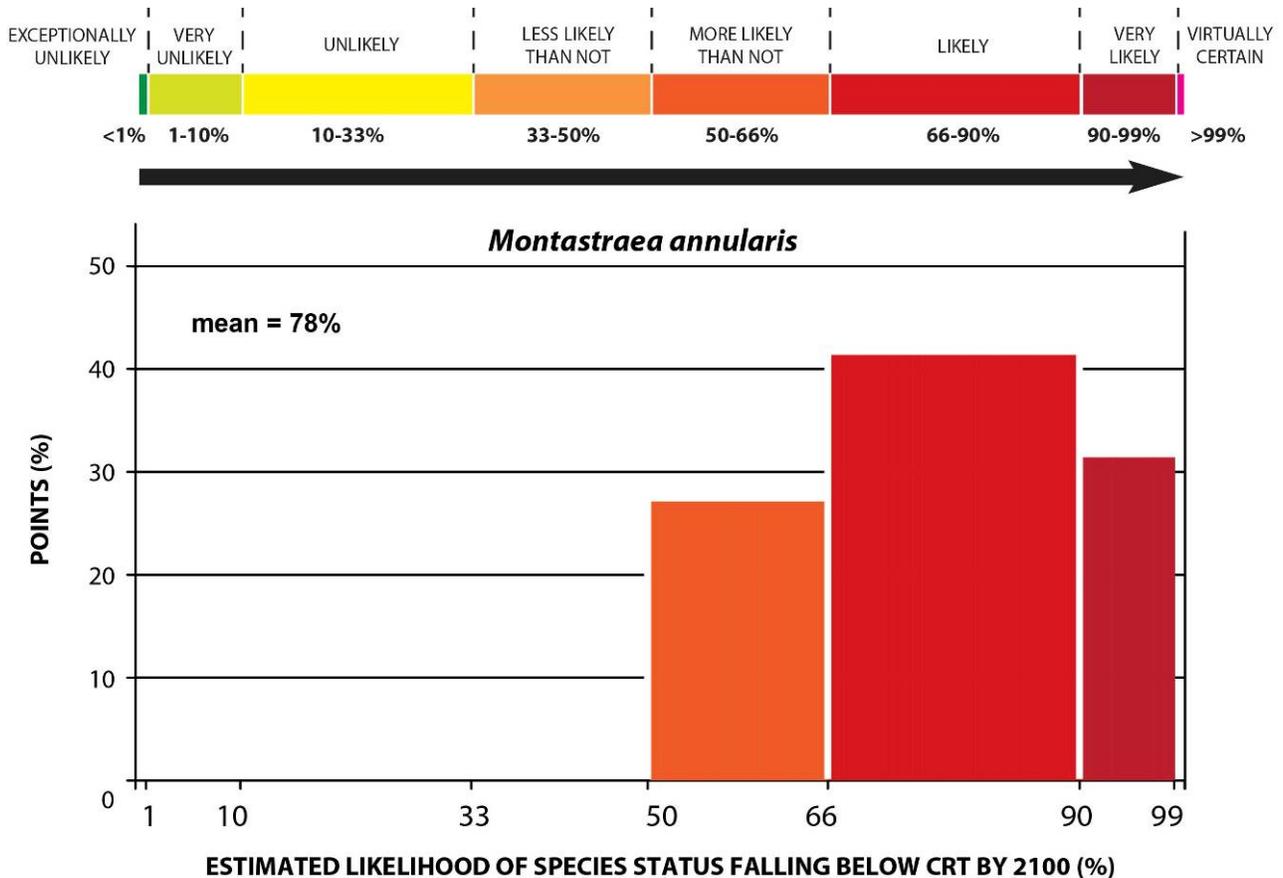


Figure 6.5.10. Distribution of points to estimate the likelihood that the status of *Montastraea annularis* falls below the Critical Risk Threshold (the species is of such low abundance or so spatially fragmented or at such reduced diversity that extinction is extremely likely) by 2100.

Factors that increase the potential extinction risk (higher likelihood of falling below the Critical Risk Threshold) for *Montastraea annularis* include very low productivity (growth and recruitment), documented dramatic declines in abundance, its restriction to the degraded reefs of the wider Caribbean region, and its preferential occurrence in shallow habitats (yielding potentially greater exposure to surface-based threats). All these factors combine to yield very high estimated extinction risk.

The overall likelihood that *Montastraea annularis* will fall below the Critical Risk Threshold by 2100 was estimated to be in the “likely” risk category with a mean likelihood of 78% and a standard error (SE) of 7% (Fig. 6.5.10). This SE was calculated by taking the standard deviation of the seven mean voting scores of the BRT members and shows the coherence among the BRT. The uncertainty of the BRT is reflected in the range of votes of 50%–99% (Fig. 6.5.10) and the average range of likelihood estimates of the seven BRT voters (45%). This overall range of votes (spanning only three categories) was small relative to most of the other candidate species.