

Genotypic inventory of *Acropora palmata* (elkhorn coral) populations in south Florida

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Background:

The elkhorn coral, *Acropora palmata*, is the iconic Caribbean reef- building species, and the key constructor of protective reef crest and spur structure. It features as the primary target in the [Mission Iconic Reefs](#)¹ plan and was listed under the US Endangered Species Act in 2005. Much of the emerging and emergency efforts toward coral rescue and propagation are currently focused on other species directly affected by Stony Coral Tissue Loss Disease rather than *A. palmata*. However, this species continues to display precipitous, and perhaps underappreciated, decline in Florida. This document provides a summary of current knowledge, both quantitative and qualitative, regarding the genotypic status of *A. palmata* populations in Florida as of 2019.

Upper Keys fixed monitoring plots:

The *A. palmata* population in the upper Keys has been the recipient of targeted, intensive demographic monitoring effort since 2004 which provides precise data on genotypic abundance within a fixed reef area over time and a strong basis for projecting the near term prognosis. The sampling design targeted 150 m² plots with moderate density of *A. palmata* colonies such that they could be reasonably tracked on a colony scale. Fifteen such plots were established in 2005 across five fore-reef spur-and-groove reef sites. All of the colonies in these plots were genotyped in 2006, when microsatellite tools became available, and an additional 10 plots and two reef sites were added as significant mortality had occurred in 2004-5 due to storms and disease. Each individual colony and fragment has been tracked in this fixed area over the full 15 years and this effort has provided solid, quantitative data on the genotypic status of this population. Full

¹ <https://www.fisheries.noaa.gov/feature-story/restoring-seven-iconic-reefs-mission-recover-coral-reefs-florida-keys>

methods description is available in (Williams et al. 2006; Williams et al. 2014). **No new larval recruits have been observed to recruit into this population over the 15 year study period** (Williams et al. 2008; Williams et al. 2014).

The loss of genotypes has been ongoing within this population, both in years of acute disturbance, as well as time frames when the population appeared to be stable or even increasing (e.g., 2006-2010; Williams et al. 2014), dropping by 64% over the course of the study (Table 1). Worse still, half of the extant genotypes persist as only remnant ‘scraps’ of tissue which are not capable of gametogenesis and spawning. Using a conservative size cutoff (40 cm) for the minimum ramet size for reproductive genets, the number of reproductive genets is only 9 of these 18. No new genotypes have been observed to recruit into this fixed area over the 15 years of monitoring. **Based on this data, it may be presumed that *A. palmata* is already functionally extinct and may be projected to local extirpation of this fixed population within 6-12 years** (Fig. 1) if the trend is presumed to be linear (constant rate of decline). However, the **rate of decline has accelerated** since 2014; carrying forward this recent rate of decline implies that extirpation may occur even sooner. This pattern is consistent with the hypothesis that this population has entered a depensatory phase, implying that continued decline is more likely than recovery. One mechanism of depensation in Caribbean *Acropora* spp. that has been long reported in the literature (Knowlton et al. 1990; Baums et al. 2003), and observed currently underway in Florida remnant populations is the concentration of corallivorous snails on remnant tissue patches (upper keys: Williams unpubl data.; lower keys: Bruckner pers comm).

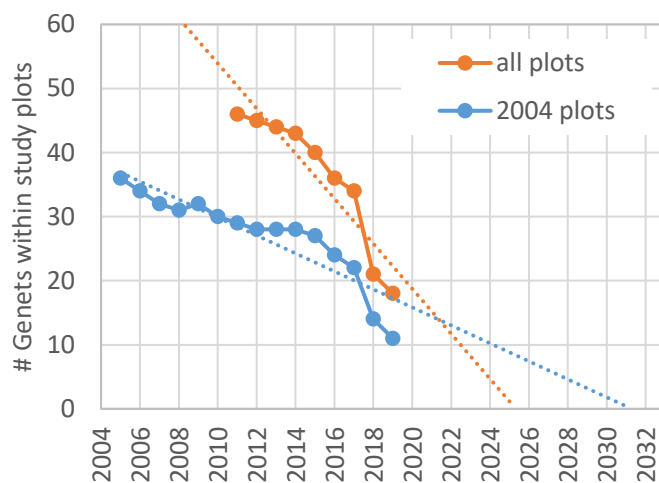


Figure 1: Trend of *A. palmata* genotype abundance within fixed monitoring area of permanent plots in the upper Keys initiated in 2004. Additional plots were established in 2010 so two trend lines are depicted (original 2250 m² for blue line; 3750 m² for orange line). Full genotypic census conducted in 2006 and in 2010 with all individual colonies and fragments tracked annually in between. Dashed lines represent projection based on linear trend (potentially conservative given the accelerated rate of decline since 2014).

Other Florida reef areas:

The above is based on a set of fixed plots in the upper Keys (Carysfort to Molasses reefs). However, there is no reason to suppose that this trend is not representative of the south Florida population as a whole. Additional targeted *A. palmata* demographic monitoring plots established

by FWC, have documented similar declines in live tissue abundance over the recent past additional sites in the lower Keys (5 sites) and Biscayne National Park (2 sites). FWC plots at Rock Key and Western Sambo have been completely extinguished while the Sand Key plots have only 2 colonies remaining (Ellis and Ruzicka, pers comm). The remaining plots showed catastrophic declines with Hurricane Irma and minimal (3 sites) increase in live cover over the two following years whereas the FWC plots at Looe Key have showed continued decline in live cover (Ellis and Ruzicka, pers comm).

Over a similar time period, several additional efforts have sought to inventory *A. palmata* throughout the keys, involving a lot of in-water searching in appropriate habitat strata both to establish additional demographic monitoring plots as the original plots dwindled (Williams et al. 2017), and as part of the effort to populate nursery stocks in the region (starting in 2013, Nedimyer and CRF). Although these efforts did not necessarily involve genotyping all colonies encountered, the location of distinct patches observed at one point in time (and later observed either remaining or dead/gone) at different sites provides reasonable inference that a trend of genotypic decline is consistent with the picture provided by the quantitative data above.

For example, we observed and recorded a set of 73 locations throughout the Keys where nursery stock collections of *A. palmata* were made between 2013 and 2018. Parsing these sites to those at least 75 m distant from each other provides a conservative inference that these sites would represent distinct genotypes. Fifty-five of these locations (or inferred genotypes) have been re-visited since 2017 to evaluate persistence or loss of live *A. palmata* by either Nedimyer or Williams. These observations indicate that **42-67% of these inferred genotypes have been lost** (depending on the fate of the unvisited ones), as of summer 2019 (Fig. 2).

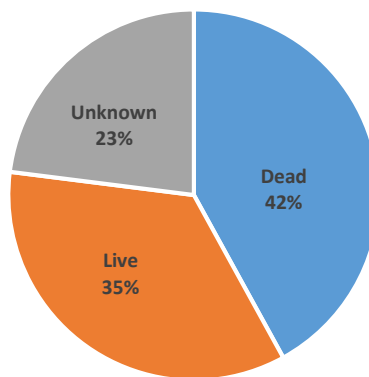


Figure 2. Proportion of keys-wide locations (>75m distant from each other) from which *A. palmata* nursery collections were made (since 2013) which have subsequently been observed as live, dead, or have not been revisited following Hurricane Irma ('unknown').

Meanwhile, additional local experts (Bartels (Mote), Bruckner (FKNMS), and Bourque (Biscayne National Park); pers comm) all confirm that the general patterns of *A. palmata* abundance throughout the lower keys and Biscayne National Park follow a similar pattern of decline as the quantitative upper Keys data. Namely, they report site-scale extirpation of formerly abundant stands has already occurred (e.g., Marker 3 reef in BNP, and with the possible exception of two colonies at Sand Key, no known native colonies persist west of Looe Key).

Prospects for the future:

It should be noted that nursery propagation and outplanting is underway throughout the Keys and patches of restored *A. palmata* provide some offset for the loss of *A. palmata* biomass (i.e. tissue area). However, they represent a subset of the extant ‘wild’ genotypes present in 2014 (i.e., when major nursery stock collections began). We believe there are approximately 120 wild genets being propagated in Florida Keys nurseries². This single pool of wild genotypes are now being populated across all restored sites, whereas in wild *A. palmata* populations, genotypes are restricted to one site. There are also some novel *A. palmata* genotypes ($n \cong 350$) that have been produced in Florida from sexual propagation and lab culture of gametes but these are largely from two cohorts (2013 and 2017; despite similar effort annually from 2013-2019) and only ~10 parental genets (K. O’Neil and E. Muller, pers comm). While the number of novel genets far exceeds the number of remaining ‘wild’ genets under propagation, many are likely siblings or half-siblings, and not all will prove fit in propagation and outplanting as they are just now being integrated into the restoration pool. Selective processes are expected to winnow these to a smaller number in both the field nursery and reef environments.

Based on two decades of *in situ* spawning observations we know that *a*) all genets do not spawn at the same time (Miller et al. 2016), and *b*) all genets are not equal in their ability to fertilize and generate viable planula (Miller et al. 2018). While a grouping of 5-6 genets in close proximity could be expected to generate larvae, their asynchrony and incompatibility mean that in practice the number needed is much larger. At present, there are no known patches of *A. palmata* that include adequate numbers of synchronous spawning genets to yield viable larvae without intervention. The successful batches produced in 2013 and 2017 include parent genets that no longer exist and the remaining genets are located at sites that are far apart (e.g., Elbow Sand Island, Turtle Rocks) making it difficult to cross fertilize even by boat.

While producing novel genets is becoming increasingly difficult, it is becoming increasingly important. The recently released ‘Mission Iconic Reefs’ plan for high profile restoration at selected reefs calls for the restoration of over 100,000 additional mature *A. palmata* colonies across 5 reefs. The resulting genotypic diversity can be considered at different scales. If we consider a keys-wide scale, 100,000 colonies fragmented from, optimistically, 470 genets (~120 ‘wild’ genets plus ~350 novel sexual recruits), this results in genotypic diversity (Ng/N) two orders of magnitude below the Acropora Recovery plan target (i.e., 0.005 compared to recovery plan target of a range-wide average of 0.5). However, it is possible that the Florida population, being latitudinally marginal, never had a genotypic diversity level near this range-wide average. If we

² This estimate is based on the following field nursery inventories: 83 at CRF, 48 at Reef Renewal, 9 at Mote Marine Lab, and 6 at University of Miami (Levy, Bartels, Baker, pers comm) and suspected overlap between nurseries of 26 genets.

instead consider site-scale diversity, at Carysfort Reef, where the 2006 baseline of Ng/N was 0.107 (Williams et al. 2014); planting 60,000 fragments from 470 genets will still result in genotypic diversity (Ng/N = 0.0075) below this baseline by more than an order of magnitude. While the number of genotypes needed to achieve this baseline (>10,000) may not seem feasible at this time, it is important to recognize the deficiency of diversity currently available. More genets of *A. palmata* are needed and producing them will likely require novel approaches in larval rearing and nursery propagation.

Table 1. Tally of *A. palmata* genotypes found within fixed study plots by reef site ($n = 3-5$ plots per reef; total area ~ 3370 m²) in the upper Keys. All colonies and fragments genotyped in 2010 and individually tracked annually since then. Half of the extant genets persist with only small remnants of live tissue which are too small to be reproductive (i.e. no ramets persist of at least 40 cm diameter or 0.16m² Live Area Index).

Reef	2011		2019		Nursery Stock ⁺
	all sizes	reproductive size	all sizes	reproductive size	
Carysfort	4	3	1	1	0
Elbow	11	7	7	5	4
French	7	5	3	1	2
Grecian	3	3	2	0	2
Key Largo DR	5	4	1	0	2
Molasses	7	5	1	0	2
Sand Island	7	5	2	1	1
Turtle Rocks	1	1	1	1	0
Total	46	33	18	9	13

⁺ indicates the number of genets found in study plots at each site that have also been incorporated into CRF nursery propagation and are presently being outplanted at additional sites.

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