Hawaii Division of Aquatic Resources

Maui and Lanai Monitoring Report

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(Includes monitoring data from 1998-2015)

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Benthic Survey Methods

The Maui benthic surveys are a collaboratively project with the Coral Reef Assessment and Monitoring Program (CRAMP) designed to gather and analyze coral data and integrate it into the overall DAR and UH-CRAMP databases. CRAMP monitoring sites (Figure 1, Table 1) were selected on the basis of existing historical data, degree of perceived environmental degradation and/or recovery, level of management protection, and extent of wave exposure. A total of 10 sites are surveyed, with two reef area stations, a shallow (1-4m) and a deep (6-13m) station at each site (Table 1).

Each station consists of ten randomly chosen 10m permanent transects marked by small stainless steel stakes at both endpoints. Digital stills photographs were taken every half meter perpendicular to the substrate at a height of 0.5m along the transect line. Approximately 24 overlapping still photos are acquired and approximately 11 non-overlapping images analyzed with Photo grid 1.0 software, for each 10 m long transect line. The analysis uses 25 randomly generated points per image with the analysis results calculated for percent benthic coverage. A detailed explanation of the CRAMP survey methodology is described by Jokiel et al. (2004a).

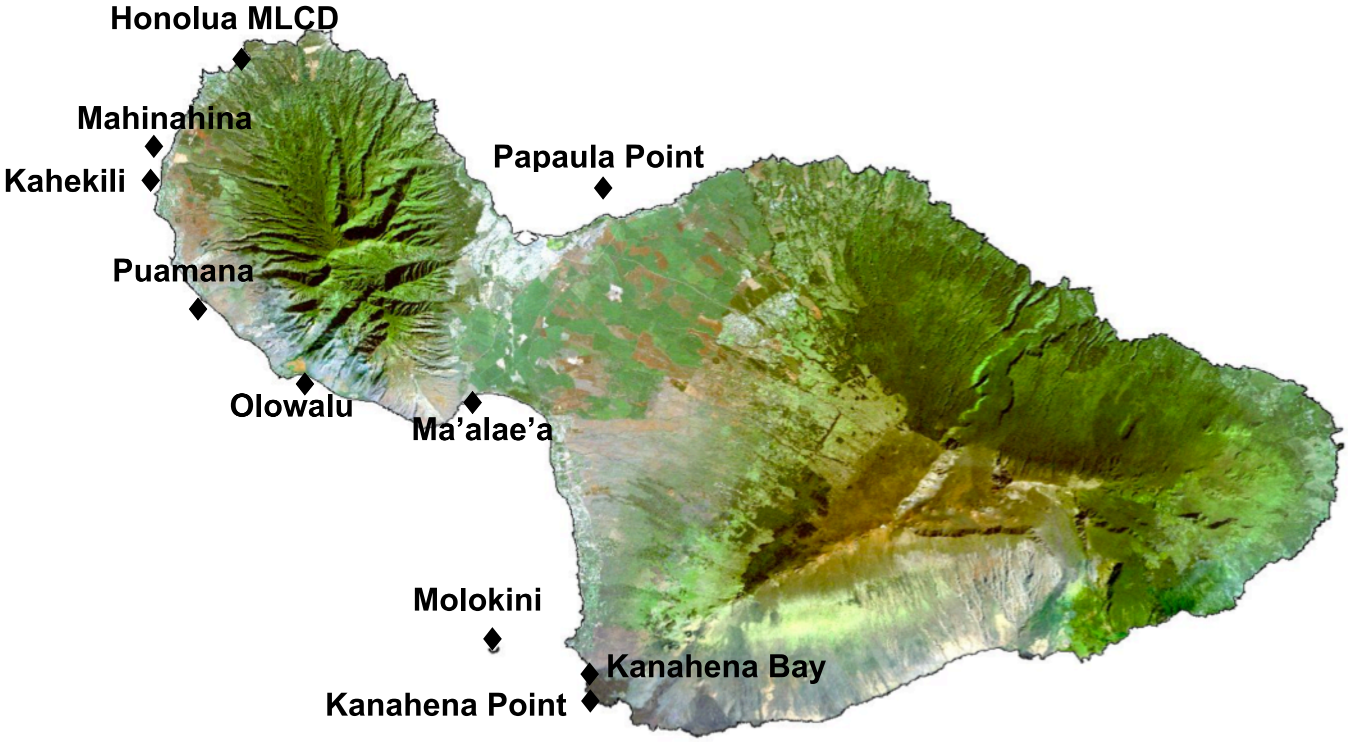


Figure 1. Maui Coral Reef Assessment Monitoring Sites

Table 1. Maui Coral Reef Assessment Monitoring Sites listed with their corresponding depth, location, and management status.

Island	Site Name	Depth (m)	Latitude	Longitude	Status
Mau'i	Honolua North	3	21.00.923	-156.38.343	MLCD
Mau'i	Honolua South	3	21.00.831	-156.38.380	MLCD
Mau'i	Kahekili	3	20.56.257	-156.41.595	OPEN
Mau'i	Kahekili	7	20.56.274	-156.41.623	OPEN
Mau'i	Kanahena Bay	1	20.37.049	-156.26.241	NARS
Mau'i	Kanahena Bay	3	20.37.015	-156.26.301	NARS
Mau'i	Kanahena Point	3	20.36.089	-156.26.214	NARS
Mau'i	Kanahena Point	10	20.36.070	-156.26.280	NARS
Mau'i	Ma'alae'a	3	20.47.378	-156.30.607	OPEN
Mau'i	Ma'alae'a	6	20.47.332	-156.30.596	OPEN
Mau'i	Mahinahina	3	20.57.436	-156.41.252	OPEN
Mau'i	Mahinahina	10	20.57.461	-156.41.336	OPEN
Mau'i	Molokini	8	20.37.889	-156.29.795	MLCD
Mau'i	Molokini	13	20.37.940	-156.29.783	MLCD
Mau'i	Olowalu	3	20.48.505	-156.36.693	OPEN
Mau'i	Olowalu	7	20.48.363	-156.36.733	OPEN
Mau'i	Papaula Point	4	20.55.307	-156.25.571	OPEN
Mau'i	Papaula Point	10	20.55.462	-156.25.571	OPEN
Mau'i	Puamana	3	20.51.369	-156.40.033	OPEN
Mau'i	Puamana	13	20.51.322	-156.40.111	OPEN

Benthic Survey Results

In 1999 and 2000, the years Maui benthic surveys started, coral cover averaged 30.7% ± 6.8% SE for the 18 stations (9 sites) around Maui County. At the same 18 stations in 2015, coral cover was 26.8% ± 7.2% SE. This slight decline in living coral cover does not appear to be ecologically significant when viewed as a whole, but the overall trend masks substantial changes occurring at individual sites. Figure 2 shows the temporal changes at the 20 currently monitored reef survey stations. Twelve of these 20 currently monitored reefs have experienced significant changes (paired t-tests of first survey year vs. most recent survey year, $p<0.05$ or $p<0.10$). Coral cover has declined at 8 sites and increased at 4 of these sites. Of particular concern are the coral cover declines at Honolua Bay, Maalaea Bay, and Papaula Point. These locations all appear to be affected by anthropogenic impacts such as land based pollution and overfishing. Conversely, sites which have sustained high coral cover, tend to be away from urban areas - either fairly remote or located offshore (Kanahena Bay, Olowalu, and Molokini). The three sites showing significant increases ($p<0.05$) are within fully protected marine reserves. The increased coral cover at these sites likely reflects natural recovery from past physical disturbances prior to the establishments of the monitoring sites. Given that these locations are away from heavy land based anthropogenic impacts and maintain high fish biomass levels (based on their no take management status), these reefs may be more resilient to infrequent disturbances like large waves, crown of thorns predation, or coral bleaching and/or disease events. Certainly, the full recovery of coral cover at Kanahena Point following a large-scale crown of thorns starfish bloom in the summer of 2005, demonstrates the high resilience at this location.

Benthic data has also been collected at six integrated fish survey sites, beginning in 2007. These data were collected on an approximately three-year cycle from Hulopoe Bay and Lighthouse point on Lanai, and at Kapalua Bay, Keoneoio bay and Keawekapu on Maui. The Canoe Beach site was added in 2013. Results of these benthic assessments are shown in figure 3. Overall there appears to be little change on these sites over the course of the six-year monitoring period. There is a marginally significant ($p<0.10$) decrease at the Kapalua Bay site and increase at the Lanai Lighthouse site. However, with only three data points, it is difficult to determine if these changes are ecologically meaningful and/or representative of longer-term trends on these reefs.

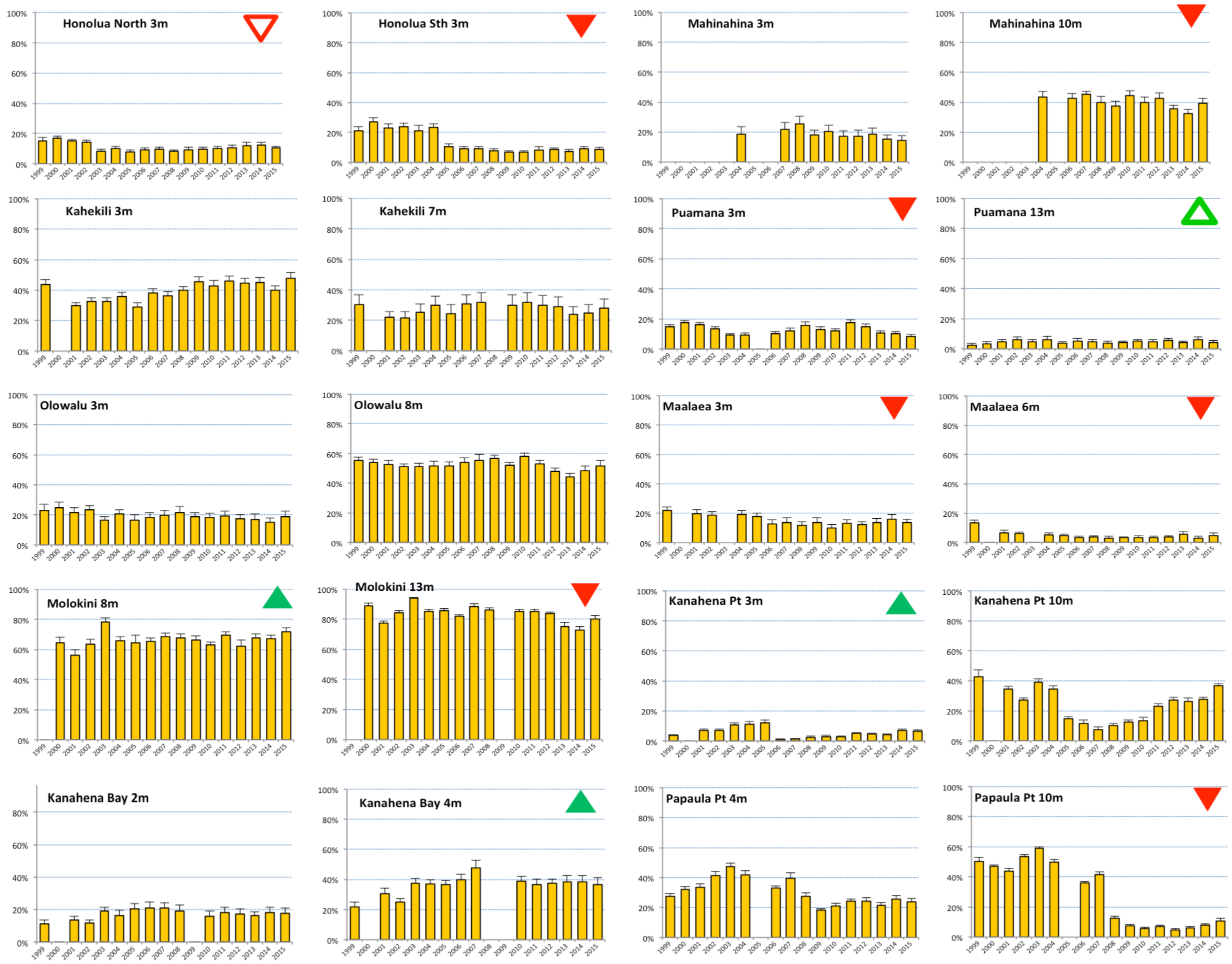


Figure 2: Temporal changes in percent coral cover at the 20 monitoring stations. Data shown as mean ± SE. Significance tests (paired t-tests) compared the first and the last year's coverage. Solid red (decrease) and green (increase) triangle represent significant changes (p-value <0.05), and open red (decrease) and green (increase) triangles represent marginally-significant changes (p-value <0.1)

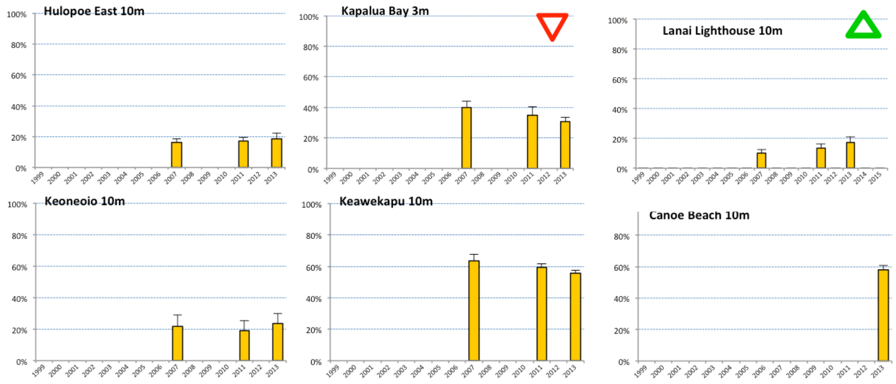


Figure 3: Temporal changes in percent coral cover at the 3 integrated fish monitoring stations. Data shown as mean. Significance tests (paired t-tests) compared the first and the last year's coverage. Open red (decrease) and green (increase) triangle represents marginally-significant changes (p-value <0.1)

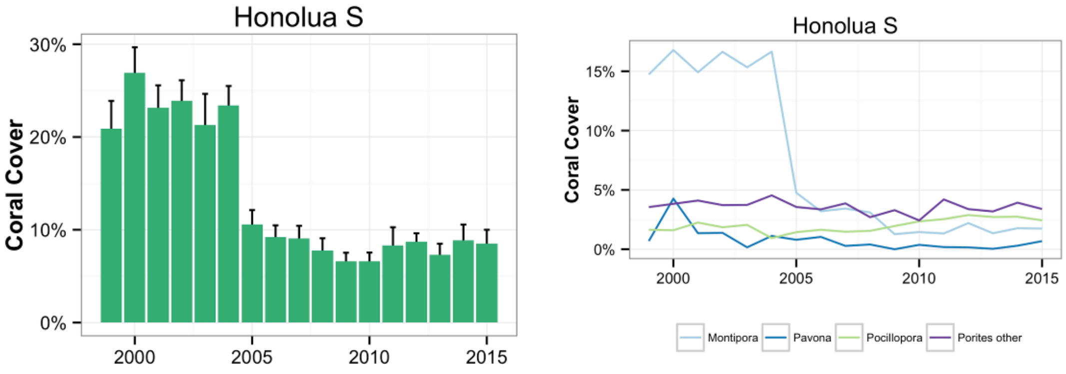


Figure 4: Honolua Bay South Reef percent coral cover plotted over time, and also shown plotted over time by individual coral genera.

The negative impacts of terrigenous sediments on coral reefs are considered a major contributing factor to reef degradation across the world (Wilkinson 2004). The coral cover within the reef flats of Honolua Bay has substantially declined over the last 15 years. This decline appears, at least in part, to be the result of large, periodic, heavy sedimentation events. A significant example was the heavy rainfall event that occurred in January 2005, which resulted in a large sediment plume within the bay. That year coral cover on the Bay's south reef declined by more than half (Figure 4). Winter swells in this bay normally flush sediment off of the reef flats, but with this particular event the water was relatively calm which allowed sediment to settle out and directly impact the corals on the southern reef flat. Further evidence that this sedimentation event led to the observed coral declines is that nearly all of the impacted coral were a shallow water species (Purple Rice Coral, *Montipora flabellata*) that is known to be relatively intolerant to sedimentation stress (Figure 4). A more recent event in May 2015 was also observed to deposit a lot of sediment and organic material into the bay. This occurred about two weeks prior to the coral survey conducted in June 2015. The recent slight dip in coral cover found at both the south and north survey sites in 2015 (figure 2) could be the result of that particular sedimentation incident as well. Large sedimentation events are a fairly common occurrence within Honolua Bay and are therefore the most likely explanation for the overall downward trend in coral cover found within the bay, as well as, the lack of any evidence of coral recovery from earlier events.

Several of the monitored Maui reefs may be experiencing negative impacts from land-based nutrient pollution. This is of particular concern for reefs with declining coral cover accompanied by increases in macroalgae cover. Maalaea Bay and Papaula Point have experienced the most severe declines (Figures 2 & 5). Similar coral declines but at a lesser extent have also been documented on reefs that have experienced periodic macroalgae cover in excess of 10% at Kahekili, Honolua, and Puamana.

In 1972, the coral reefs within Maalaea Bay were described as being "striking in their diversity and in the presence of rare corals species" (Kinzie, 1972). Similarly, a U.S. Fish and Wildlife environmental assessment in 1993 estimated coral cover in the vicinity of the current CRAMP survey stations to be between 50% and 75% (USF&W, 1994). These scientific assessments describe a once healthy and diverse reef ecosystem. The Maalaea reef is now extremely degraded and has experienced periods of heavy algal overgrowth. The recent decline in macroalgae at these Maalaea reefs maybe partially related to a rapid increase in the abundance of the collector urchin (*Tripneustes gratilla*) (Figure 5). Collector urchins were observed moving into the area in large numbers in 2010. It is not clear, what lead to this large migration of adult collector urchins into the shallow reefs of Maalaea, but they were scarce prior to that period and have been documented in fairly high abundance ever since.

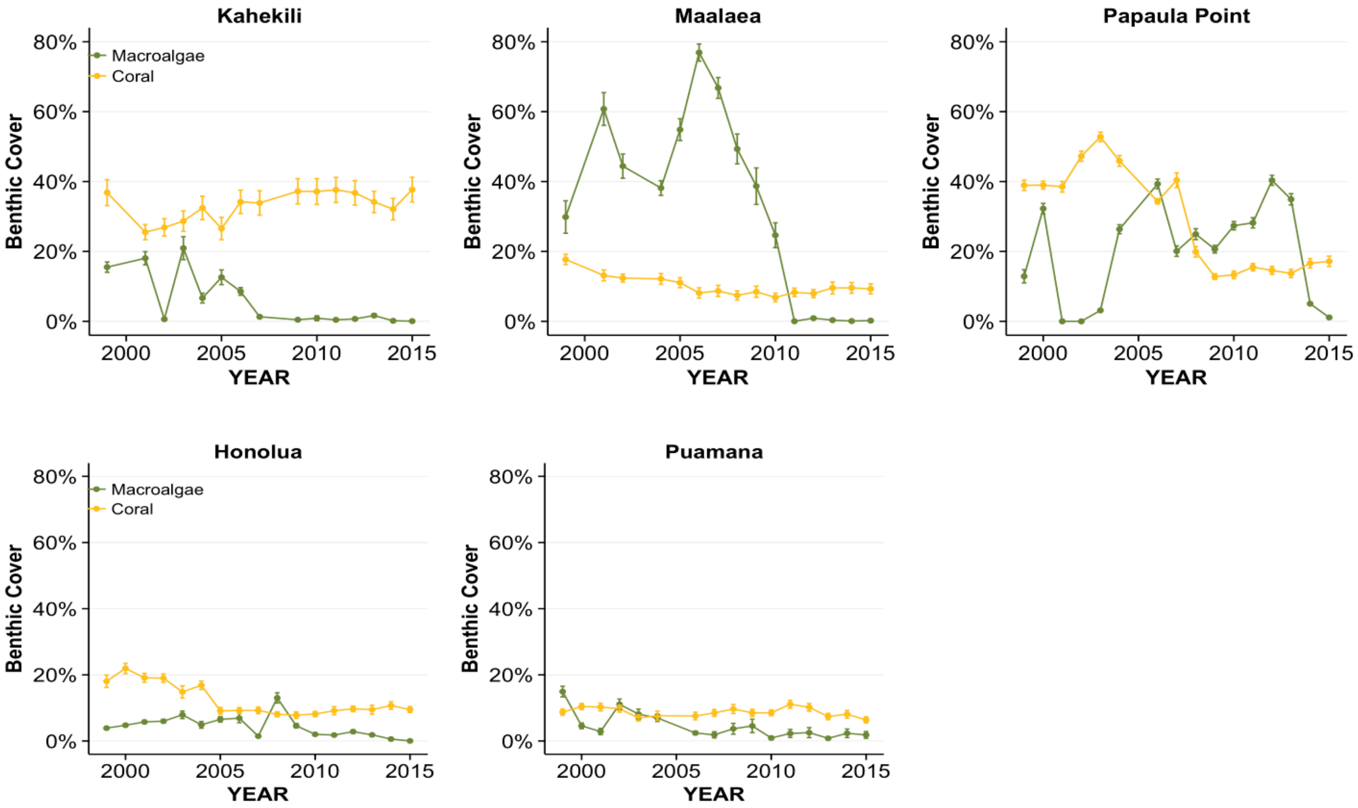


Figure 5: Long-term temporal changes in percent coral cover (yellow line) and macroalgae (green line) at the Kahekili, Maalaea, Papaula Point, Honolua Bay and Puamana survey sites (graphs include both the shallow and deeper stations combined at each location).

At Papaula Pt, coral cover on the 10m station declined from around 50% in 2002 to less than 10% in 2015. Much of this decline occurred between 2007 and 2008 (Figure 2 & 5). In addition, there has been a consistent and dramatic increases in macroalgae, particularly *Acanthophora spicifera* at this station starting in 2004 and remaining high through 2013. More recently, however, the macroalgae cover at this location has greatly declined for unknown reasons (Figure 5). This recent and substantial decline in macroalgae has been observed at reef sites all over Maui, but it is not currently clear what factors may be contributing to these observed island wide changes in macroalgal abundance.

At both the Maalaea and Papaula monitoring sites, the loss of coral cover has effectively resulted in ecologically significant coral reef ecosystem degradation, with fish stocks apparently suffering the double whammy of overfishing and lack of suitable habitat. These sites now support relatively few herbivorous fishes, and that fact, along with elevated nutrient levels, likely contributes to the observed high macroalgal cover at these locations in some years. Elevated nutrients have been implicated as a causal factor in *Hypnea* and *Ulva* blooms at other areas around Maui (Smith and Smith, 2006). However, in the case of *Acanthophora spicifera*, which is a highly preferred food for grazing fishes (Hunter, 1999), low grazing pressure might be a more fundamental causal factor. There appears to be a relationship between highly-depleted herbivore stocks (e.g. Maalaea) and abundant *Acanthophora*, and conversely, no or very limited *Acanthophora* growth at sites where grazing fishes are abundant (e.g. Honolua Bay).

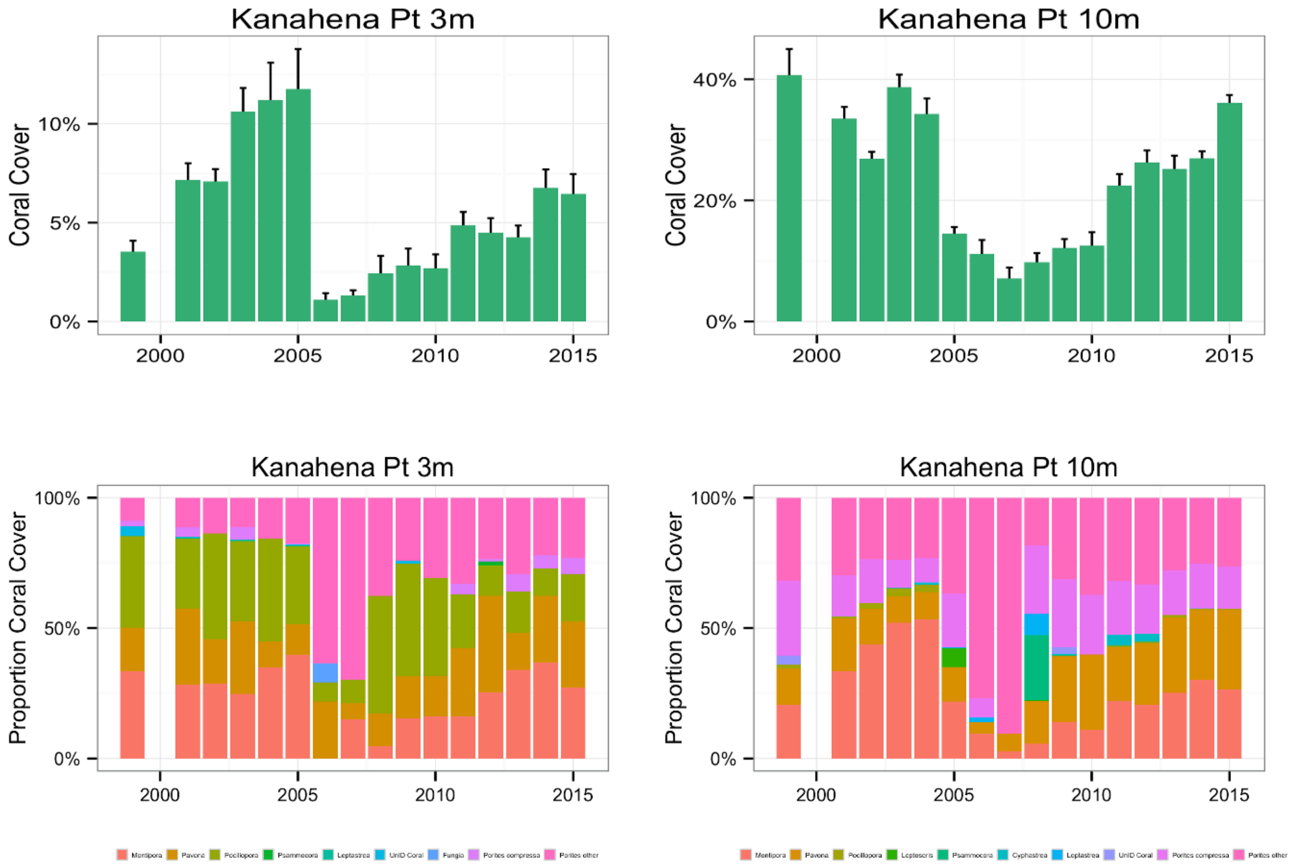


Figure 6: Coral percent cover for Kanahena Point 3 & 10m monitoring stations plotted over time. Different color bars (lower graphs) represent different coral genera plotted over time.

Finally, the dramatic decreases in live coral cover at the Kanahena Pt 10m station in 2005, and Kanahena Pt 3m station in 2006 (Figures 2&6) were caused by a localized outbreak of the crown of thorns starfish (COTS). Increased COTS densities were initially observed in areas just southeast of Kanahena Point in 2004. At the time of our surveys in 2005, COT density was roughly one starfish per 10 m^2 of reef at Kanahena Pt. Before the COTS outbreak, coral cover on the deep station was at 34.4% in 2004, and at 11.9% on the shallow station in 2005. After the outbreak, coral cover dropped to as low as 7.4% at the deep station in 2007 and 1.1% at the shallow station in 2006. The most affected coral genera was *Montipora*, whereas other genera, particularly *Porites*, appeared to be much less affected (Figure 6). Fortunately these reefs appear to have recovered rapidly. A comparison of the coral cover on the deep station from 2007 to 2015 showed a significant increase (paired t-test $p<0.01$). On the shallow station, comparisons between 2006 and 2015 show a similar recovery (paired t-test $p<0.01$). Although the COTS outbreak caused a sharp decline in coral cover, the reefs at Kanahena Point appear to have bounced back to levels comparable to those recorded prior to the COT predation event (2015 mean cover being 6.5% at the shallow station and 36.9% at the deep station). It is too early to determine if this localized coral predation event will result in long-term changes in coral diversity, but short term there was a clear change from montiporid towards more poritid corals (figure 6- 2006 & 2007) and after one decade post event, there appears to be a fairly equal distribution of the main coral genera. Overall increase in coral diversity within a reef system could help make the reef more resistant to future stressors and improve overall resilience (McClanahan, et. al. 2012; Carpenter 1997; Birkeland and Lucas 1990).

Mau‘i Fish Surveys

Fish Survey Methods

Three types of fish surveys are conducted on Maui: (1) resource fish surveys, (2) “integrated” fish population and urchin surveys, and (3) nearshore habitat and fish assessments (HAFA).

Resource Fish Surveys

The resource fish surveys are conducted three times per year at eight sites (Figure 7). Four sites that are within a reserve, where fishing is prohibited or severely restricted and four sites are within a ‘control’ area, being somewhere close by with relatively similar reef structure and where fishing is permitted. The area pairs are:

Three marine reserves on Mau‘i:

- Ahihi-Kinau Natural Area Reserve (NAR), control at La Perouse Bay
- Molokini MLCD, controls at Makena and Keawakapu
- Honolua-Mokuleia MLCD and control sites between Kapalua Bay and Lipoa Point

One marine reserve on Lanai :

- Manele-Hulopo`e MLCD and control area ‘Lighthouse; on southwest coast of Lanai.

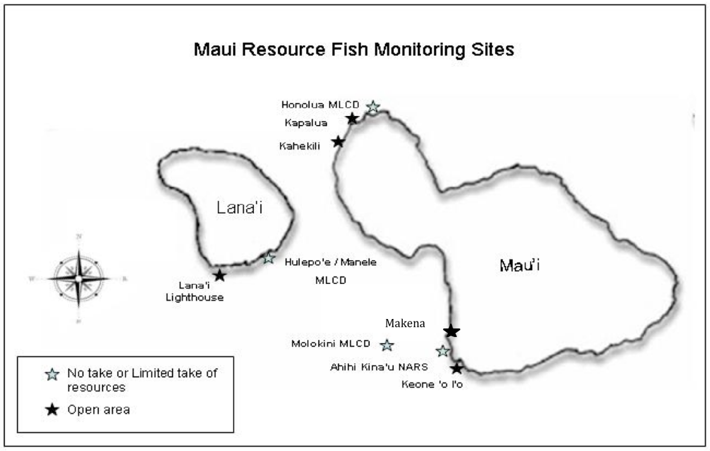


Figure 7. Maui resource fish survey areas

All of the Maui closed areas are fully protected no-take reserves. Manele-Hulopoe MLCD on Lanai permits pole and line fishing from shore. The Maui County survey areas were selected to allow pairwise comparisons between protected areas and controls, but it should be noted that differences in habitat and exposure exist between reserve and control areas, particularly between the Molokini MLCD and the Ahihi-Kinau NARs and their respective control areas. Therefore, although we attempt to draw conclusions about performance of individual reserves, we also look for broad patterns across all areas within each management strata (protected vs. open).

Five sub sites are surveyed per survey location (3 at 3-5m depth and 2 at 10m depth) using the ‘resource fish’ survey method. For this method, 2 pairs of divers start at a fixed center point and head in opposite directions. Each pair of divers swim parallel to each other, 10m apart, and follow a depth contour, for a five minute period. Each diver records all main fishery target species that are ≥ 15cm and within a 5m wide belt. Beginning in 2007, changes were made to include fish (> 10 cm) for two species of smaller surgeonfish (Achilles tangs (*Acanturus achilles*) and Manini (*Acanthurus triostegus*)). All other fish species continued to be surveyed as before. Dives are conducted using SCUBA. Abundance and biomass are then standardized for the area covered on each timed swim.

Starting points for each survey are based on the site coordinates for the center point. End points are determined by taking a GPS point from a Garmin handheld GPS that is attached to the dive float.

Integrated Fish Surveys

An 'integrated' fish survey, wherein all fish species and select invertebrates are assessed, was also conducted at each of the eight locations. Integrated survey sites used the H transect design shown in Figure 8. This design is consistent with the fish survey methods in West Hawaii and will allow for comparisons to be made with this data. Six stainless steel eyebolts (the circles in Figure 8) permanently mark the location of the four 25m transect lines.

Starting at the central eyebolt, two divers swim side-by-side on each side of the line, each survey a 2 m wide belt. This creates a 4 m wide belt transect. On the outward-bound leg, each diver rolls out their respective transect lines while counting and sizing all fish ≥15 cm (TL) within the full 4 m wide transect. On the return leg, fishes closely associated with the bottom, juveniles, and fishes hiding in cracks and crevices are recorded and sized. The sizes of all fishes are visually estimated to the nearest 5 cm and recorded in 5 cm bins (i.e. 1-4.9 cm="A", 5-9.9 cm="B", 10-14.9 cm="C", etc.). Measured hash marks on the top of diver-held data slates serve as visual size references. Fishes whose sizes indicate they have recently recruited are noted as "R". After the detailed fish assessment, the dive pair swims back towards the end pin recording the number of large sea urchins within a 1 m swath on their side of the line. Each diver pair conducts surveys along two 25 m lines that start at their central eyebolt. When finished with both lines, each diver conducts a 5-minute present/absent survey of all fish species that were not seen on transect, but were in the general area of the survey site.

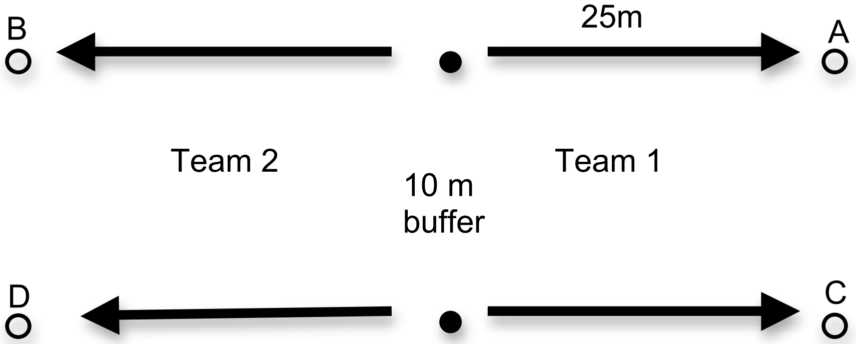


Figure 8. Diagram of 'integrated' fish survey transect configuration. Solid dots represent the center and the hollow dots the designated endpins.

HAFA Surveys

The nearshore habitat and fish assessment or HAFA surveys are designed to record both the abundance and size of targeted fish species to establish the status and trends of specific reef fishes that were commonly taken by lay-gillnets prior to the lay-gillnet ban that went into effect in March of 2007 (HAR 13-75-12.4). Since several of the fish species potentially affected by gillnet fishing are herbivores, the surveys also assess relative sea urchin abundance, and benthic cover [e.g., coral, crustose coralline algae, macroalgae, sand, and substrate (rock, rubble, turf algae, etc.)].

HAFA Surveys are conducted three times per year at seven shallow water reef areas where lay-gillnets were previously used (Figure 9). Each survey location has around eight sub-sites.

A HAFA survey is comprised of two parts: (1) an outward swim while counting fish and by (2) a fixed point benthic assessment conducted every minute on the outward swim. Swimmers start their swim at a fixed GPS point. A five-minute rapid assessment swim on a designated bearing is utilized with one pair of observers (snorkeling) at a depth contour of 2-4m. Within a 5m wide belt, each observer records all herbivorous fishes ≥ 10cm as well as, all other resource fish (wrasse, goatfishes, snappers, etc.) ≥ 15cm. The benthic assessments are conducted at the start of the transect and then again at each one-minute swim interval. This process results in a total of six benthic assessments per transect. The benthic assessment is a qualitative assessment of benthic cover conducted by looking at an estimated 5m-radius circle of benthos centered directly below the surveyor. When the five-minute fish survey is complete, the GPS location is marked providing a measure of the total distance covered by the transect. Upon completion of the fish and benthic survey, the surveyors rank the urchin and general algae abundance using a DACOR scale (Dominant, Abundant, Common, Occasional, or Rare). In addition, the dominant algal species are identified and recorded.

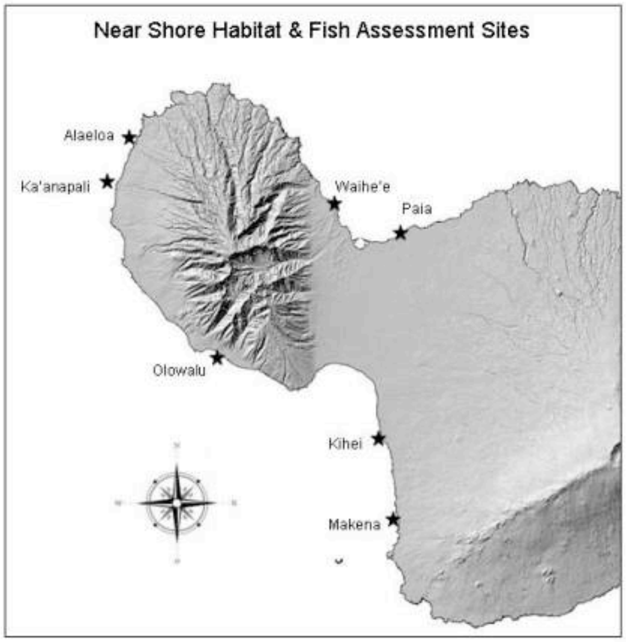


Figure 9. Maui HAFA survey sites

Data Analysis

For all fish surveys the total length (TL) was estimated to the nearest 5 centimeter size bin. Length estimates of fishes from visual censuses were converted to weight using the following length-weight conversion: $W = aL^b$. The parameters a and b are species-specific values for the allometric growth equation where L is fish length in mm and W is weight in grams. In cases where best available length-weight conversion parameters were for standard length or fork length (SL or FL), lengths in TL were first converted to FL or SL. These length-length and length-weight parameters were obtained from FishBase (http://www.fishbase.org) and from unpublished data on 150 species commonly observed on visual fish transects in Hawaii (Hawaii Cooperative Fishery Research Unit). In the cases where length-weight information did not exist for a given species, the parameters from similar congeners were used. All biomass estimates were converted to grams per square meter ($g\ m^{-2}$) to facilitate comparisons with other studies in Hawaii.

Fish Survey Results

Resource Fish and Integrated Surveys

Overall, the fish survey results indicate a positive effect of closure to fishing. Compared to their controls, two of the fully closed reserves (Honolua-Mokuleia MLCD, and Molokini MLCD) had higher total resource fish biomass levels (Figure 10, all resource fish graph, $p<0.05$ paired t-tests). There was no clear significant difference between Ahihi-Kinau and Hulopoe Lighthouse when compared to their controls, but there was a slight trend towards higher biomass in the control areas at both of these locations. As previously noted, there are important differences in habitats and exposures between some of the reserves and their associated control areas. This is particularly true between the Ahihi-Kinau NARs and La Perouse Bay. In La Perouse Bay, large sand patches and channels surround the reef structure. This abundance of sand habitat near the surveyed coral reefs likely contributed to the high biomass of some large schooling surgeonfish and goatfish (e.g. *Acanthurus olivaceus* and *Mulloidichthys flavolineatus*) at that site. Those species tend to be associated with the specific habitats found in La Perouse Bay, which likely was a factor in the significantly higher biomass of mullids and acanthurids there compared to Ahihi-Kinau (Figure 10). It is also noteworthy that the highest fish biomass at any surveyed area was at the Lanai Lighthouse control location (a fished, but fairly remote location). It would therefore be overly simplistic to ascribe all differences among areas simply to management status.

The high fish biomass at the Lanai sites and the lack of a clear distinction between the partially closed (Manele-Hulopoe MLCD) and the open area (Lighthouse) merits further comment. Lanai has a small resident population, and as a result, sites there are likely to have lower fishing pressure than most reefs on west Maui. Also, the Manele-Hulopoe MLCD is the only Maui county reserve area that is not a complete no-take reserve – as fishing with pole and line from shore is permitted. The MLCD also has the most accessible section of coastline on the island, with a paved road leading down to the ocean and a public park with showers and bathroom facilities. Because of the easy access to Manele and Hulopoe Bays, it is likely that even though spear fishing, netting, and vessel-based fishing are prohibited, this area still gets a large portion of near-shore fishing activity in that vicinity. In contrast, the Lighthouse control area is located along the southwest coast of Lanai, where the shoreline is only accessible via a rough off-road trail. Fish behavior and shore-based structures indicate that the Lighthouse area is fished, but it seems likely that fishing pressure is relatively low. It therefore seems plausible that there is little real difference in fishing pressure between the Lanai reserve and open areas.

Parrotfish, which are highly targeted, were found to have significantly higher biomass in all four reserve locations compared to their controls ($p<0.05$), and apex predators (emperors and jacks) showed a clear trend towards higher biomass levels within all the fully protected reserves (Ahihi-Kinau NAR, Molokini MLCD, and Honolua MLCD). These differences were significant at the Ahihi-Kinau NAR and the Molokini MLCD when compared with their respective controls ($p<0.05$) (Figure 10).

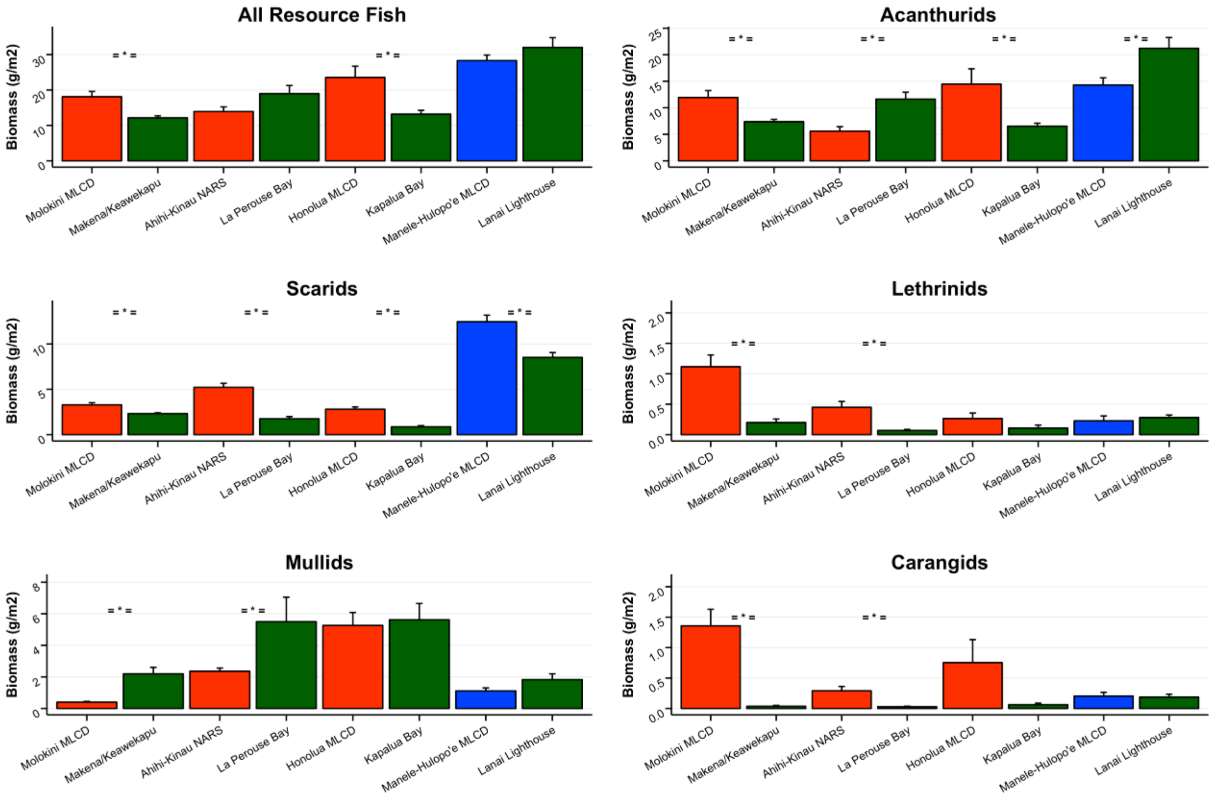


Figure 10. Mean and SE of resource fish biomass levels (g/m^2) at DAR Maui County monitoring stations. Data are averages of all surveys in 2011-2015. Significant pair-wise differences between an MPA and control area is indicated by "=*=" ($p<0.05$). Red bars are areas where no fishing is permitted, green are areas outside of reserves and blue indicates where pole and line fishing is permitted, but other forms of fishing are prohibited.

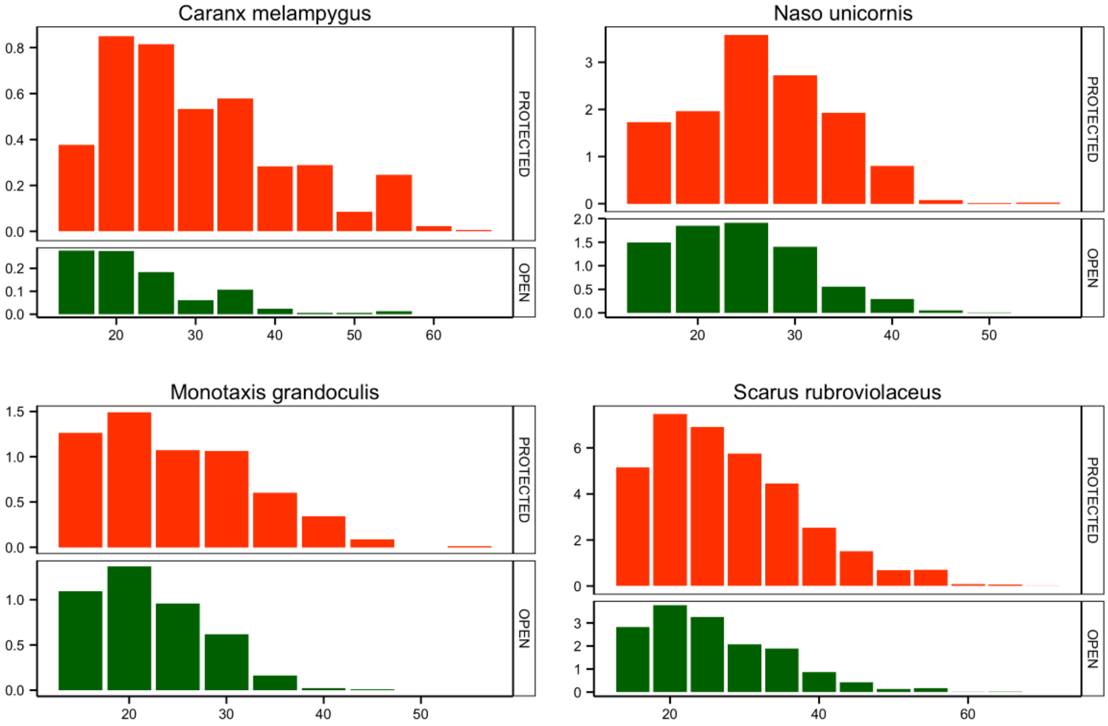


Figure 11. Number and sizes of key target fishes in protected and open areas on Maui. Data pooled into all protected monitoring stations (red) and all open sites (green).

Size distribution trends were investigated for four relatively commonly encountered and heavily targeted fish species [*Caranx melampygus* (Bluefin Trevally), *Naso unicornis* (Bluespine Unicornfish), *Monotaxis grandoculis* (Bigeye Emperor), and *Scarus rubroviolaceus* (Redlip Parrotfish)]. For all four species, reserves contained more and larger fishes than open areas (Figure 11). The biological implications of these results is important, as larger individuals tend to be a very important component of species' breeding stock. They produce disproportionally more gametes than smaller fish, and those gametes tend to be more able to survive to become recruits (Birkeland & Dayton, 2005). Marine reserves make up less than 2% of nearshore waters in Maui County; which clearly limits their potential for substantially increasing spawning stocks across the County as a whole. Nevertheless, our results indicate that these few marine reserves likely contribute disproportionately to fish breeding stocks.

In order to investigate the effectiveness of past fisheries management efforts, the data were analyzed for any apparent trends in fish biomass (figure 12). Lay-gillnets were banned as a fishing method in all Maui waters in 2007. Since net fishing is an effective way to capture large quantities of schooling surgeonfishes [e.g. *Acanthurus triostegus* (convict tang / manini), *Acanthurus olivaceus* (Orangeband surgeonfish / Naenae)], and schooling goatfishes [e.g. *Mulloidichthys flavolineatus* (yellowstripe goatfish / wekea) and *Mulloidichthys vanicolensis* (yellowfin goatfish / weke ula)], it might be expected that that banning the use of this fishing method would result in increases in the biomass and/or abundance of these types of fish species. As shown in figure 12, significant increases in surgeonfish occurred at La Perouse Bay and in goatfishes at Makena (90% CI). Goatfishes also appear to be trending towards higher biomass levels at two other open fishing areas (La Perouse and Kapalua Bays). Although it is too early to tell for sure, the fact that these increasing trends were only apparent in open fishing areas and not within existing marine reserves, where lay-gillnetting was already prohibited, provides some evidence that the 2007 lay-gillnet ban could be having a positive effect on biomass of schooling fishes at some heavily fished locations.

Low parrotfish biomass and the lack of any positive trends at Maui sites compared to Manele/Hulopoe MLCD and Lanai Lighthouse (figure 12) indicates that this ecologically important group of fishes are probably depleted around Maui. This and other information along with concerns expressed by local stakeholders including fishers and the general public led to the implementation of new parrotfish regulations in November 2014. These new rules ban the harvest of terminal phase parrotfishes of the two largest species and set a two fish total daily bag limit on all parrotfishes. Given the life history of these fishes (including lifespans of ~20 years for the largest bodied species), it will take considerably more time before we will be able to judge the effectiveness of those management actions.

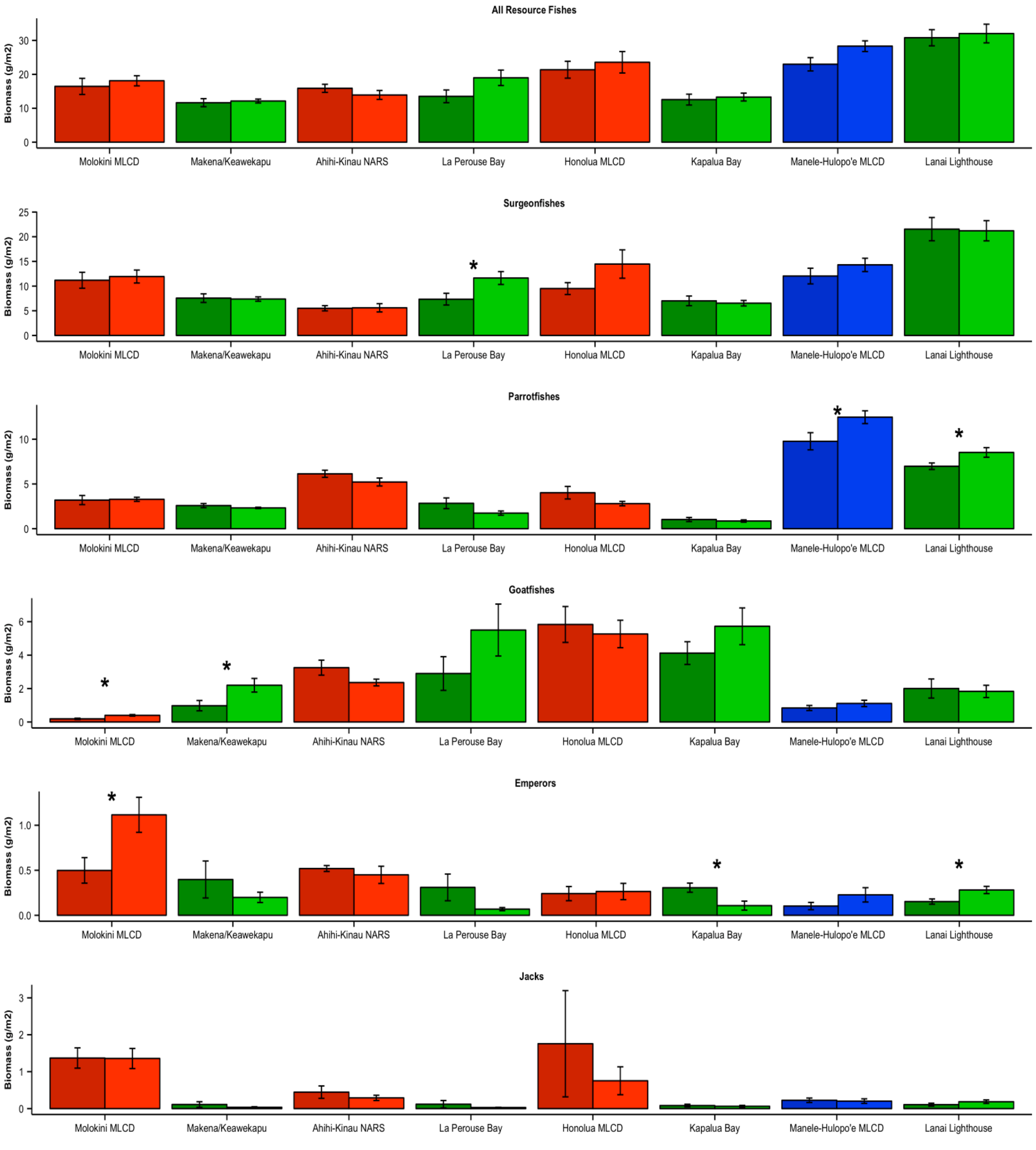


Figure 12. Mean and SE of resource fishes biomass levels (g/m^2) at DAR Maui County monitoring stations. Data are shown as comparisons between the averages of all surveys in the 2006-2010 period and the 2011-2015 period. Significant pair-wise differences between time periods at the same location are indicated by * (90% confidence intervals, i.e. equivalent to p<0.1). Bar colors represent management status and time period; dark red (no-fishing 2007 - 2010), lighter red (no-fishing 2012 - 2015), dark green (open to fishing 2007 – 2010), lighter green (open to fishing 2012 - 2015), and dark blue (partial protection 2007 - 2010) and lighter blue (partial protection 2012 - 2015).

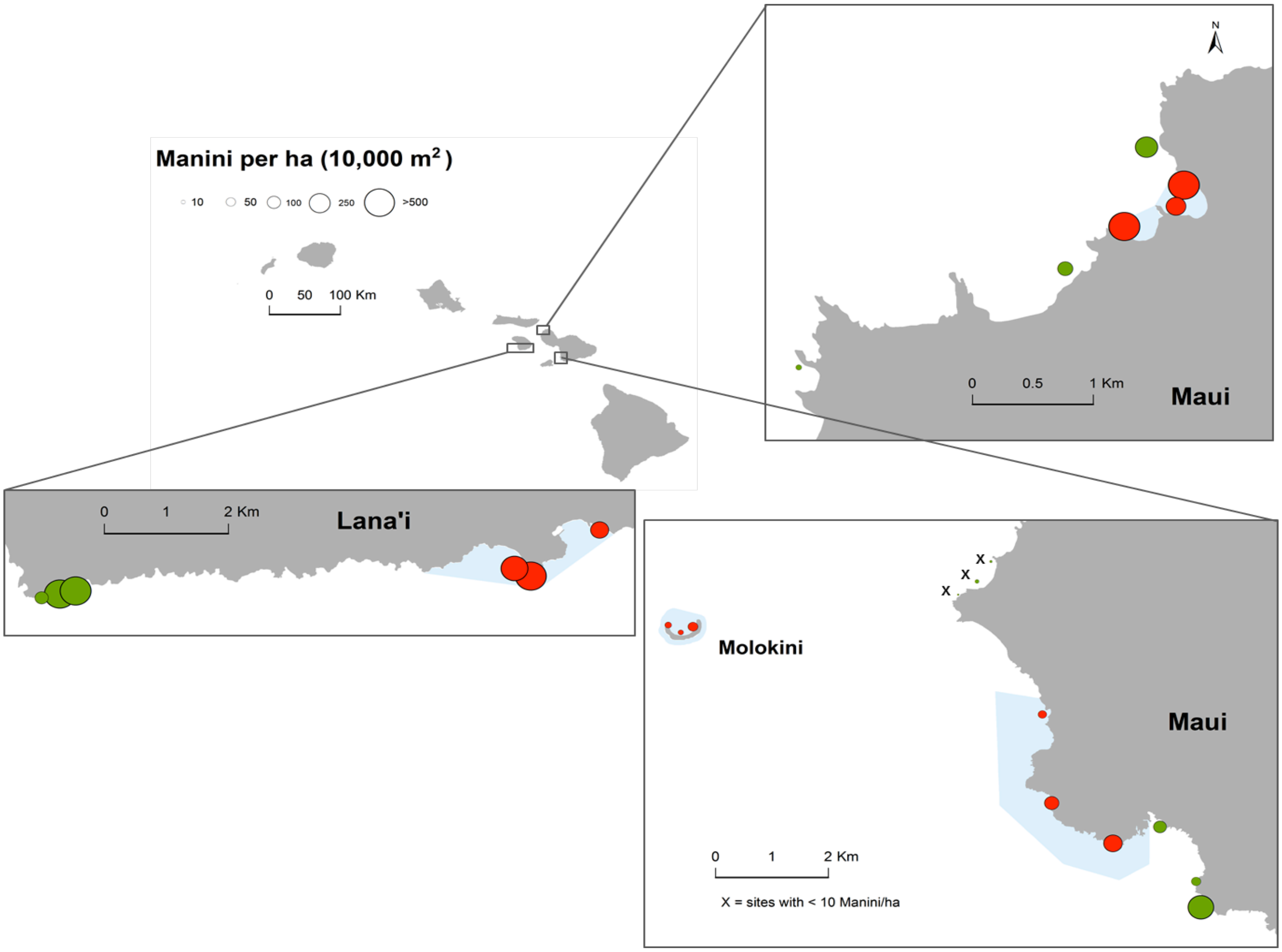


Figure 13. Manini abundance (#/m^2) at Maui DAR monitoring stations. Density represented by size of circles. Red circles are areas where fishing is either restricted (Lanai) or prohibited (Maui), green circles are from open fishing sites.

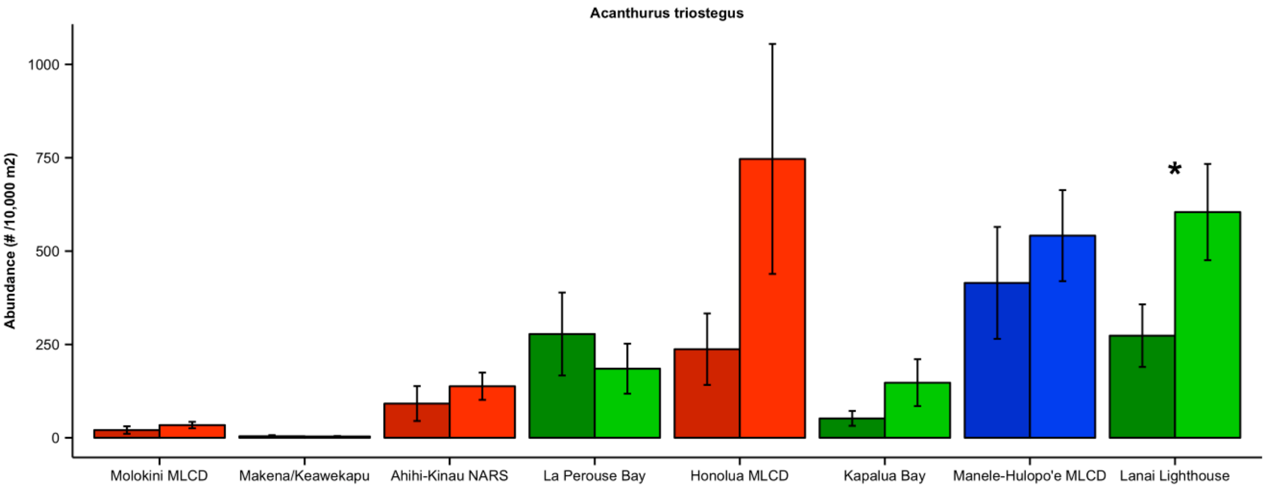


Figure 14. Trends in manini (*Acanthurus triostegus*) abundance (#/hectare) at all the Maui DAR monitoring stations. Significant pair-wise differences between time periods at the same location are indicated by * (90% confidence intervals). Bar colors represent management status and time period; dark red (no-fishing 2007 - 2010), lighter red (no-fishing 2012 - 2015), dark green (open to fishing 2007 – 2010), lighter green (open to fishing 2012 - 2015), and dark blue (partial protection 2007 - 2010) and lighter blue (partial protection 2012 - 2015).

Overall review of the Maui County fish surveys indicate that herbivore fish stocks are depleted at many of the survey locations. Several areas (Figure 10) had low acanthurid and scarid biomass levels. Large schools of manini (*Acanthurus triostegus*), a key shallow-water grazer, were mostly limited to reserves or in the relatively remote area on Lanai (Figure 13). Both Lanai areas had large populations of manini, but the only survey sites on Maui Island with abundances greater than 500 manini/ha, occurred within the Honolua MLCD. In contrast, within the last four years, all of the Maui open access sites had manini densities below 250/ha except for one location on the far southern point of La Perouse Bay (Figure 13). These patterns suggest that fishing pressure has impacted the abundance and distribution of this species.

Trends in manini abundances from the four-year period (2007 – 2010) versus the most recent period (2012 -2015) show that, with the exception of La Perouse Bay, manini abundance appears to have increased at all the study sites (figure 14). However, it is important to understand that because manini are schooling fishes, there is inevitably a large degree of variability in biomass estimates, resulting from chance encounters with schools of several hundred fish. For example, the largest increase in manini abundance occurred at the Honolua MLCD site, but 2012 was an unusual year with the observed numbers of manini found to be three times higher than the next highest year. Regardless of this high level of variability, the overall patterns suggest there have been real increases in overall numbers across all management levels (fully protected and open fishing areas). In fact, the largest increases appear to have occurred at the sites that had the highest manini abundances to begin with (both Lanai sites and the Honolua MLCD). It is most likely, therefore, that these observed increases in manini abundance are independent of management, but more likely the result of increased recruitment success or other ecosystem changes.

HAFA Surveys

The new lay gill-net regulations that went into effect in March 2007 (HAR 13-75-12.4) and which banned the use of lay-gillnets in Maui waters, were a response to concerns that lay-gillnet fishing was indiscriminately catching and killing large numbers of nearshore reef fish. Many of these fish were herbivores, which provide ecosystem services necessary for maintaining healthy coral reefs. The HAFA surveys were implemented in early 2007 in an effort to develop baseline information on shallow water reef fishes that tended to be heavily fished by lay-gillnets. These surveys looked primarily at fish assemblages, but surveyors also made semi-quantitative visual estimates of benthic cover.

In general, we found that data from the HAFA surveys tended to have high variability that made it difficult to draw statistically robust conclusions. That high variability was likely due to combination of: (i) inherent high variability of fishes in shallow water, as many of the fishes we were interested in are sometimes found in large schools; (ii) a high degree of habitat patchiness in the surveyed shallow water areas; and (iii) large impact of conditions between surveys, e.g. wave energy and turbidity both on fish distributions and on the surveyors ability to conduct surveys. Figure 15, shows a comparison of the fish biomass at the seven HAFA survey locations between the 2008-2011 and the 2013-2015 time periods. Only the Alaeloa location showed any significant difference (i.e. 90% confidence intervals of change did not overlap 0) with decreased biomass of surgeonfishes. It is not clear what drove the decline in surgeonfish biomass at Alaeloa, but there was a similar decline (although not significant) at the Paia site. Much of the surgeonfishes biomass in these shallow water surveys comes from the very abundant and mostly non-targeted schooling brown surgeonfish (*Acanthurus nigrofuscus*). It is possible therefore, that these observed changes in surgeonfish biomass reflect natural fluctuations in the abundance of brown surgeonfish. Fish biomass at the Makena location appears to have substantially increased with the 2013-15 mean more than double the 2008-11 value.

Although that change was not statistically significant (due to the high variability associated with schooling goatfishes), it mirrors the significant results discussed earlier for the resource fish surveys conducted in the Makena area (figures 10 & 12). As with the resource fish surveys, these increases in fish biomass appear to be almost completely the result of increases in the abundance of the schooling goatfish (*Mulloidichthys flavolineatus*) (figure 15, goatfishes graph). Given the fact that these goatfish are highly targeted and fairly effectively captured with lay-gillnets, these results provide possible evidence that the lay-gillnet ban may have resulted in increased biomass of certain targeted schooling species.

The qualitative nature of the benthic data gathered during these surveys makes it difficult to draw firm conclusions. However, anecdotal information provided by the survey team suggests that fairly large shifts in benthic cover occurred at some of the survey locations. For example, at some sites, there were large increases in the cover of the zooanthid (*Palythoa caesia*). This trend was most obvious in the Waihee area where zooanthid cover increased from near zero percent to well over 10% cover in a 3 year period. For the most part, however, the visual-estimate benthic assessment methods were too variable and open to observer bias to be useful in evaluating habitat changes. As a result, future surveys will be based on fully-quantitative methods such as benthic photo-transects.

The HAFA surveys were discontinued in 2015 and replaced by a rapid assessment method similar to that used in the Kahekili HFMA (see the Kahekili Herbivore FMA assessments described below). This new survey method has been implemented at numerous sites throughout the state, is currently utilized by the Nature Conservancy's marine survey team at various community locations around Maui County and was developed to carefully investigate fish biomass levels in relation to specific habitat characteristics (Friedlander, et. al., 2007). Given that this survey method involves multiple replicate fish and benthic surveys and that it provides a quantitative assessment of both fish levels and benthic composition, it should provide a much better measure of resources in the surveyed areas and therefore will greatly increase our ability to measure change allowing for more accurate assessments of both past and potential future management actions.

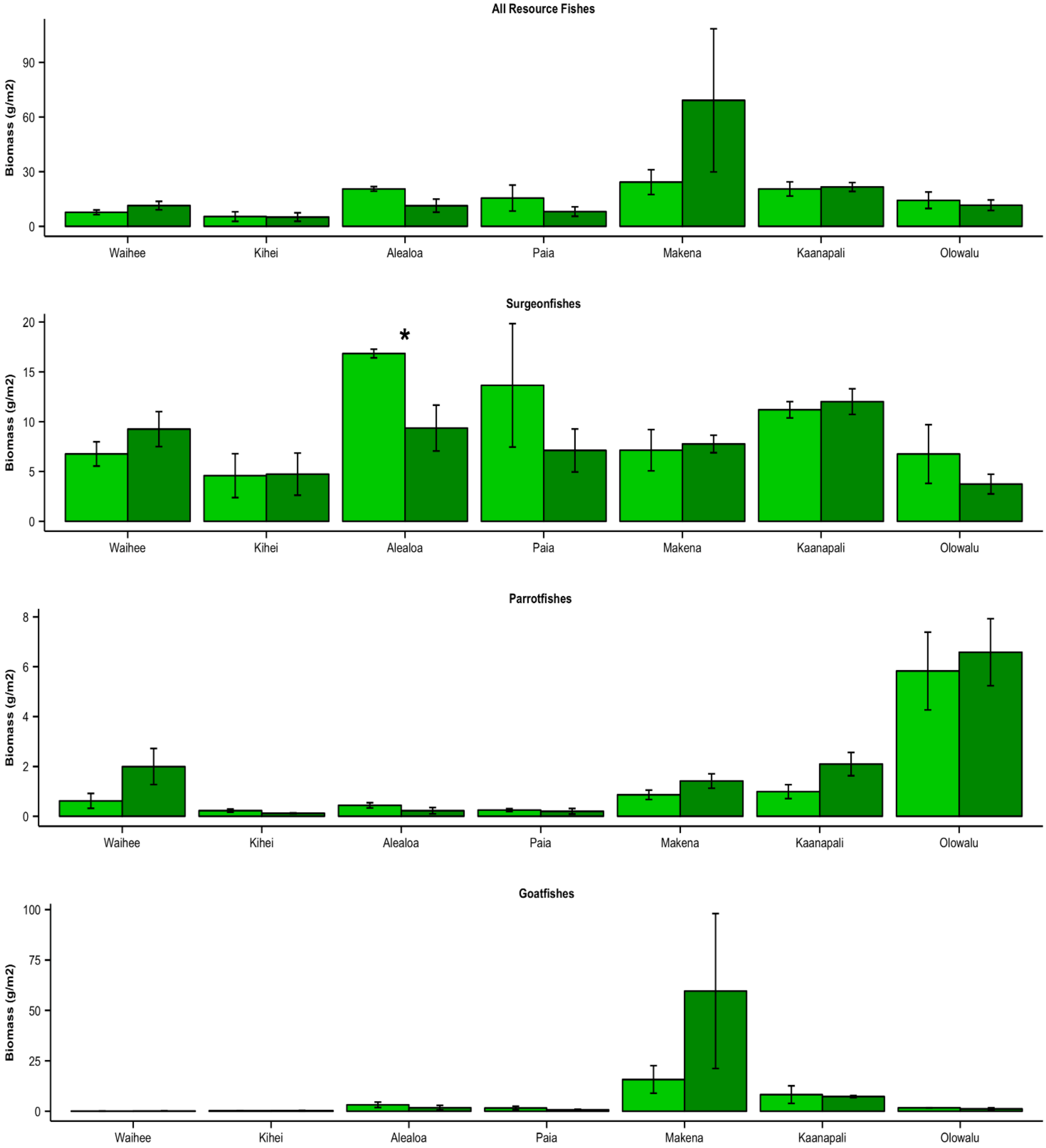


Figure 15. Mean and SE of resource fishes biomass levels (g/m2) at the seven Maui County HAFA survey sites. Data are shown as comparisons between the averages of all surveys in the 2008-2011 (light green) period and the 2013-2015 (dark green) period. Significant pair-wise differences between time periods at the same location are indicated by * (90% confidence intervals).

Kahekili Herbivore FMA Assessment

Summary

In July 2009, the Hawaii Division of Aquatic Resource (HDAR) established the Kahekili Herbivore Fisheries Management Area (KHFMA). This new marine managed area encompasses coral reefs offshore of the Kahekili Beach Park in West Maui and was implemented in November 2009. The KHFMA was created in response to concerns about reef conditions in the area; ephemeral blooms of macroalgae from at least the late 1980s (Soicher & Peterson, 1997; Smith, Runcie & Smith, 2005), and declining coral cover at nearshore survey sites near the middle of the KHFMA from ~55% in 1994 to ~35% in 2006 (Friedlander et al., 2008).

Corals tend to be negatively impacted by macroalgae and other algae (e.g. thick turfs) that tend to be abundant on reefs when herbivory is low; whereas substrates that tends to dominate in heavily grazed reef environments – i.e. bare substrate and crustose coralline algae (CCA) – are favorable for coral growth and recruitment. Thus it was believed that restoration of natural grazing process in the KHFMA would help to shift the competitive balance from algae back to corals. To that end, the KHFMA rules prohibit the take of herbivorous fish and sea urchins, but other forms of fishing are permitted.

At the time of establishment, KHFMA reefs were still in relatively good condition, thus it seemed likely that they would be able to support abundant herbivores and ultimately coral recovery if herbivore protection was effective. The KHFMA was intended as a management strategy for the reefs inside its boundaries, but also as a test of the potential wider applicability of herbivore management. This assessment effort was designed to generate meaningful data against which eventual effectiveness of the KHFMA could be assessed.

Here were present data from 8 years of monitoring, beginning 19 months prior to the establishment of the KHFMA. Monitoring has been conducted as a series of partnerships between DAR, UH Botany Department, and NOAA-Coral Reef Ecosystems Program. KHFMA hardbottom habitat was grouped into six broad habitat categories (Figure 16, Table 2), and on 1-2 occasions each year, beginning in January 2008, teams of divers have conducted haphazardly located surveys in intensive 4 day rounds of monitoring. In each round, the goal is to spread sites as widely as possible broadly covering the full extent of the hardbottom habitat and with adequate replication within each different habitat categories (see example of January and August 2008 survey rounds in Figure 16).

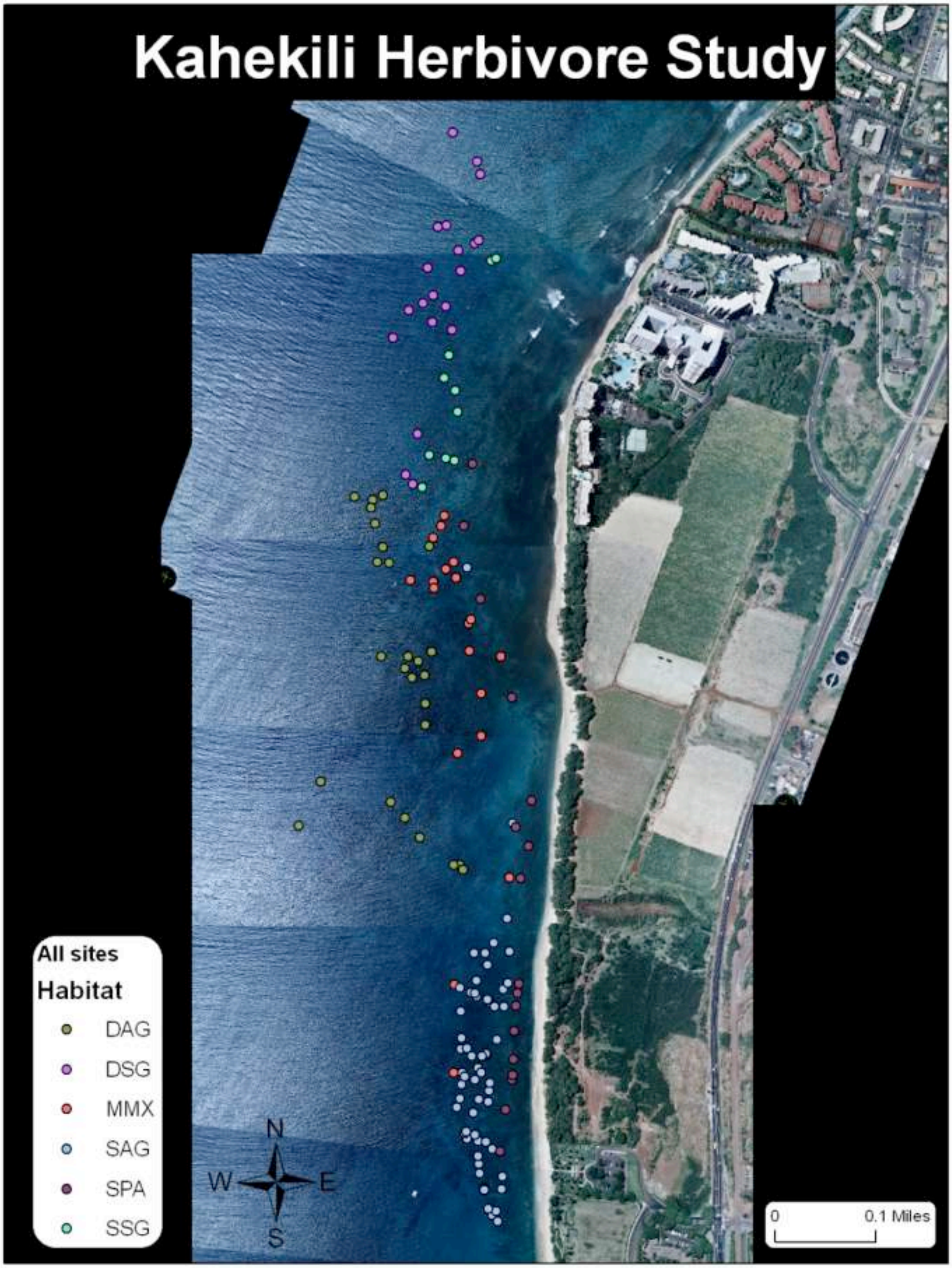


Figure 16. Location of 2008 Kahekili Baseline Surveys. January and August 2008 surveys combined. See Table 2 for a description of the six habitat types.

Table 2. Habitat classifications within the Kahekili HMFA.

Category	Description	Area (Ha) [%]
Shallow Pavement (SPA)	Largely flat, low-relief and low coral cover areas dominated by limestone pavement and loose sediment, typically ~1-5 m deep.	2.35 [8.5%]
Shallow Aggregate Reef (SAG)	Moderately or highly complex reef adjacent to shore, with moderate to high coral cover and considerable structure arising from coral growths ~2-8 m deep.	3.96 [14.3%]
Deep Aggregate Reef (DAG)	Same as SAG, but these are offshore areas of reef, generally ~7-16 m deep.	5.33 [19.3%]
Mixed Mid-depth (MMX)	Benthos generally dominated by loose sediment and sand with sparse corals, but with patches of higher coral cover, typically 3-8 m deep.	3.51 [12.7%)
Shallow Spur-and-Groove (SSG)	Shallow portion of spur-and-groove habitat, where spurs are distinct but less well developed than deeper areas (i.e. spur height generally <2.5m); depths generally ~3-5 m.	4.96 [18.0%]
Deep Spur-and-groove (DSG)	Very well developed spur-and-groove habitat, with spur heights often 3-5m or more in depths of ~4-15m.	7.50 [27.2%]

Methods

Surveys were conducted from a small boat with survey teams of two divers. The divers entered the water over hard bottom habitat and swam straight down to the nearest suitable habitat (hard bottom large enough to lay a 25m survey transect in). One diver tied the starting point of the survey transect and the other recorded the transect start location using a GPS in a waterproof bag attached to a surface float.

One diver conducts fish surveys in which the species, number and size (in 5 cm slots) was recorded for all fishes larger 15 cm total length (TL) within a 4-m wide belt centered on the diver as they laid out the 25 m transect tape. The diver would then return along the transect, recording species, number and size of all fishes smaller than 15 cm TL in a 2m wide belt centered on the transect line.

The other diver followed the fish surveyor, and conducted a photo-quadrat survey of the benthos under the transect line, and then recorded all sea-urchins within a 1m-wide belt, during the return swim along the transect line.

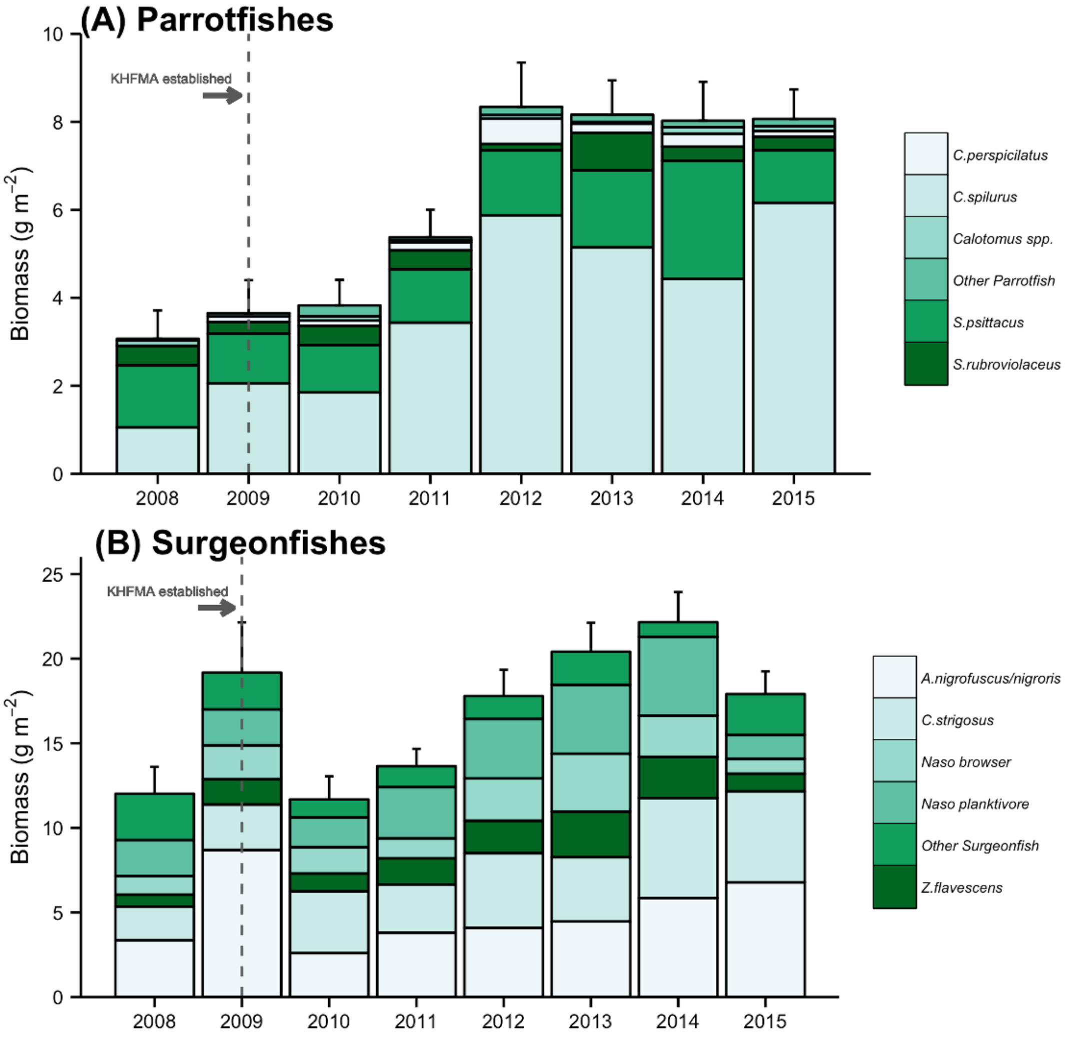


Figure 17. Trends in biomass of herbivorous fishes. Error bars represents standard error by family. '*Naso* browser' are *N. uncornis* and *N. lituratus*, and '*Naso* planktivore' are made up of *N. hexacanthus* and *N. brevirostris*.

Results

Parrotfish biomass increased rapidly after establishment of the KHFMA, approximately plateauing from 2012 onwards (Figure17). Overall, between 2008-9 and 2014-15, parrotfish biomass increased by 139 % (95% CI: 95 to 184 %). Estimated biomass of surgeonfishes was anomalously high in 2009, mainly due to encounters with large roving schools of *Acanthurus nigrofuscus* and *Acanthurus nigroris* in shallow habitats, but otherwise appeared to be trending upwards after closure, until declining slightly in 2015 (Figure 17). In spite of the high counts in 2009, total surgeonfish biomass increased by 28% (95% CI: 3-54%) between 2008-9 and 2014-15, with increases being most evident for two species, *Zebrasoma flavescens* and *Ctenocheatus strigosus*.

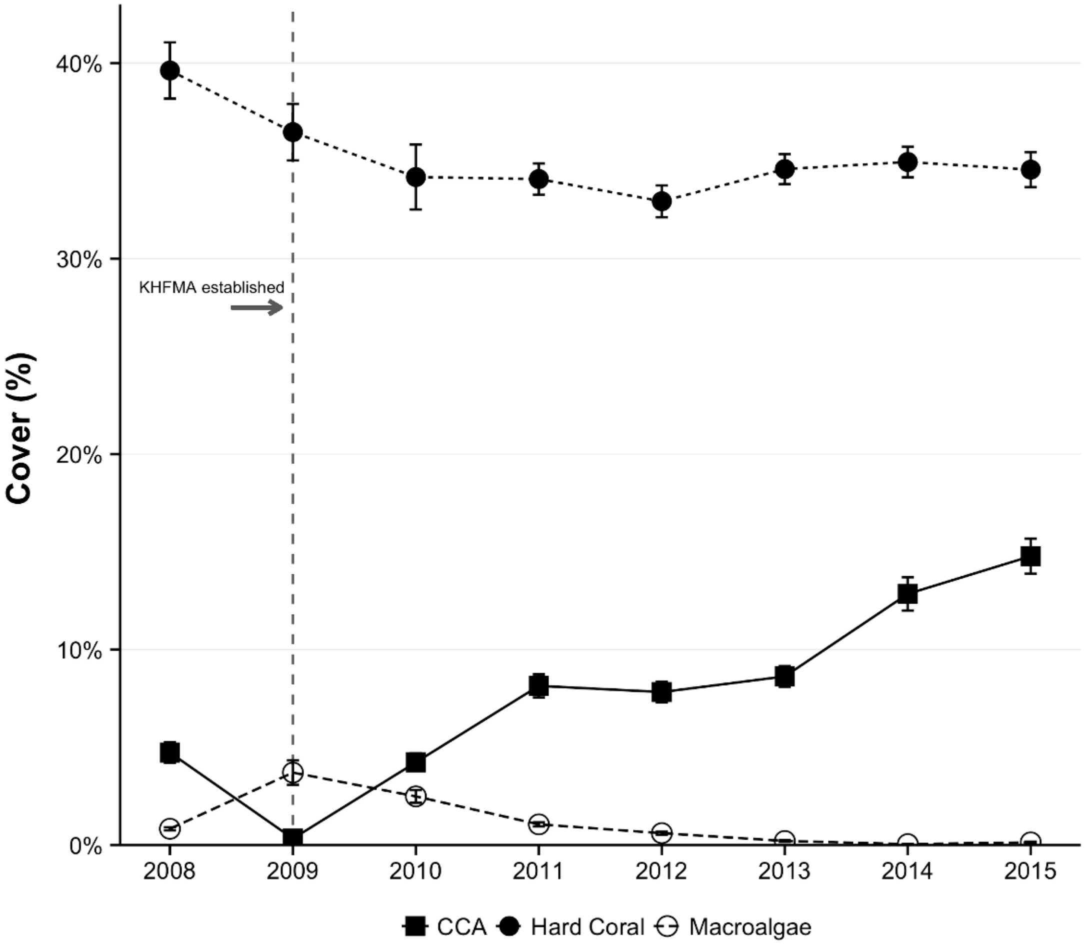


Figure 18. Trends in benthic cover at KHFMA. Data shown are annual mean and standard error.

Coral cover was declining at the time the KHFMA was established, with a clear downward trend between 2008 and 2010, and reaching a low in 2012 (Figure 18). However, coral cover subsequently stabilized, and appeared to trend upwards though 2014 (Figure 18), and the first round of surveys in 2015 (not shown). By the time of the second survey round in 2015, a major statewide coral bleaching event was well underway. Many corals in the KHFMA were bleached and there appeared to be some associated mortality, with the net effect being that mean cover for all of 2015 was slightly lower than in 2014.

Macroalgae cover was low at the time of all survey rounds, but has been virtually nonexistent in most recent years, with the highest cover in any year being 3.7% in 2009 (Figure 18). The clearest change in benthic communities since establishment of the KHFMA has been a steady and substantial rise in crustose coralline algae (CCA), which increased from 2.5% in 2008-9 to 13.8% in 2014-15 (Figure 18).

Overall, there is clear evidence that the establishment of the KHFMA has increased herbivorous fish stocks and that benthic algal assemblages have already changed in ways that are likely to favor corals. It is encouraging that the downward trend in coral cover at KHFMA has stopped and that coral cover has subsequently been stable or even increasing. However, full recovery of herbivorous fishes and potentially corals will take considerably more time. Therefore DAR and partners will continue to monitor reefs in the KHFMA

Coral Health Assessments

Ahihi-Kinau MWS Disease Outbreak

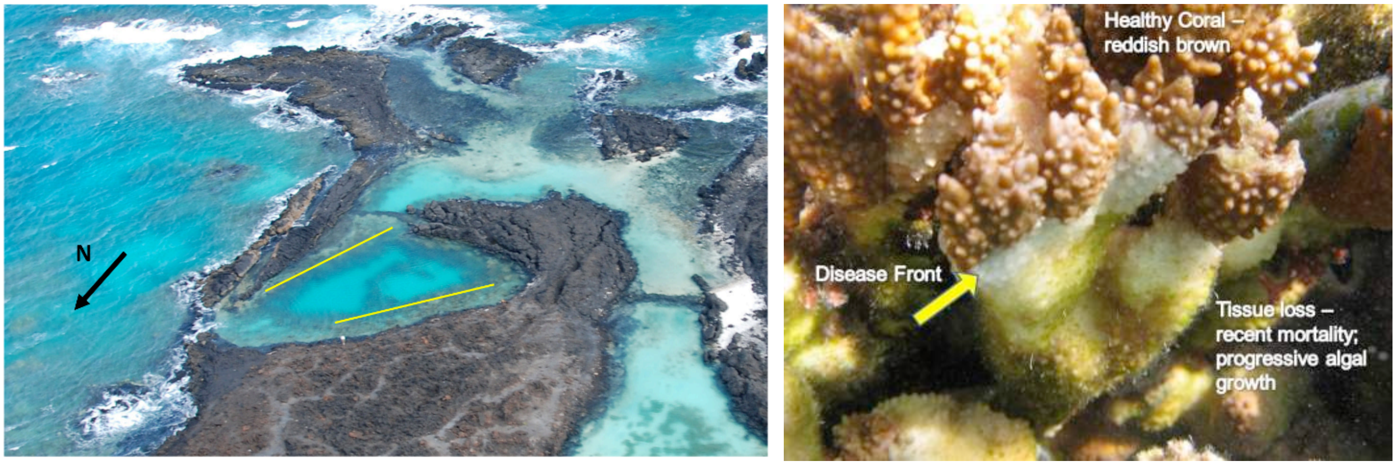


Figure 19. *Montipora* pond in the Ahihi-Kinau Natural Area Reserve System (NARS) Maui, Hawaii. The pond is ~38m in maximum diameter with two 25m transects located on the east and west edges of the pond. White Syndrome (tissue loss) outbreak at Ahihi-Kinau Natural Area Reserve, in a semi-enclosed pond adjacent to La Perouse Bay.

In July 2008, an outbreak of the tissue loss disease *Montipora* White Syndrome (MWS) was discovered in the ʻĀhihi Kīnaʻu Natural Area Reserve on the southern shore of Maui. This outbreak had been ongoing for approximately one year when DAR first learned of it. The outbreak occurred in a small semi-enclosed near shore pond that is dominated by the affected coral, *Montipora capitata.* Disease prevalence and coral cover were quantified periodically from September 2008 to March 2011. MWS prevalence ranged from 9.5% in 2008 to 1.8% in 2011. Monthly or bi-monthly surveys of ten marked colonies (eight diseased and two unaffected) using photographs and semiquantitative estimates of live, diseased, and old dead tissue were surveyed from March 2010 through March 2011. Diseased colonies observed over time showed a mean tissue loss of 40.8% ± 10.6% SE per colony per year. The mean benthic cover of *M. capitata* declined from 48.5% ± 20.9% SE to 27.5% ± 1.5% SE within two years representing a total reduction in coral cover of 43.3% ± 14.8% SE (Figure 20). This is the highest reported MWS prevalence and the first report of a significant reduction in coral cover associated with a disease outbreak in Hawaiʻi (Ross et al, 2012a). Disease prevalence declined with coral cover (Figure 20), and this decline is presumably due to reduced host abundance.

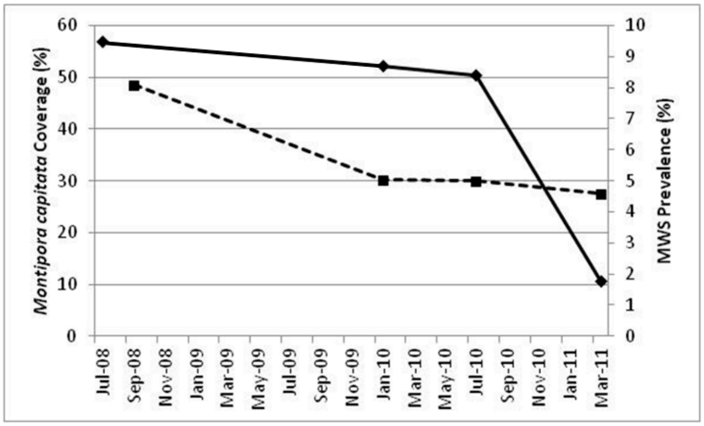
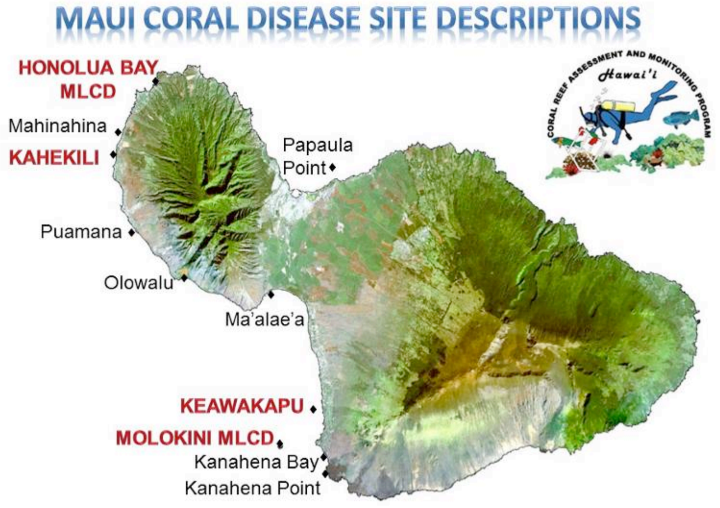


Figure 20. Mean *Montipora capitata* cover (dashed line) and *Montipora* White Syndrome prevalence (solid line) at *Montipora* pond from July 2008 to March 2011.

Coral Reef Watch Disease Surveys



The NOAA Coral Reef Watch, NOAA/NMFS Coral Reef Ecosystems Program (CREP), and researchers from the Hawaii Institute of Marine Biology (HIMB) partnered with DAR on each island to collect coral disease data to feed into a computer model for the development of a satellite prediction tool for coral disease outbreaks in Hawaii. This tool utilizes sea surface temperature (SST) data as an indicator of potential disease outbreaks, and has been thus far successful on the Great Barrier Reef. Hawai'i posed new challenges to this tool, given that the reefs are very close to the shoreline, increasing the need for finer resolution imagery. Surveys were performed three times a year (April, July, and late September) over two years (2010-2011). The four priority sites for this project included established fish survey and CRAMP sites at Molokini, Keawakapu, Kahekili, and Honolua Bay. Additionally, all of the CRAMP sites were surveyed in July 2010. These data were incorporated into the Hawaii Coral Disease (HICORDIS) database along with all other disease surveys conducted in the Hawaiian Archipelago, in order to better understand the risk to corals in the face of increasing ocean temperatures for enhanced monitoring and management. Three diseases were investigated for thermal stress anomalies: *Montipora* white syndrome, *Porites* growth anomalies, and *Porites* tissue loss syndrome. *Montipora* white syndrome was revealed to be influenced by warm water events, however, the strongest predictor was winter conditions and host density (Caldwell et al. 2016). The NOAA Coral Reef Watch satellite predictive tool for coral disease has demonstrated the ability to be a useful tool for managers in Hawai'i.

Kahekili Dead Zone Survey

The Kahekili Dead Zone Survey study was designed to map areas of degradation and to describe patterns of mortality on the coral reefs offshore of Kahekili Beach Park in Kaanapali, Maui. This site was selected based on the 36% decline of coral coverage from 1994 to 2006 (Friedlander et al., 2008), with recent evidence that sewage effluent is being introduced to the reef via injection wells (Dailer et al., 2010; Glenn et al., 2013; Bishop et al., 2015; Swarzenski et al., 2016), and observations of discrete areas of nearly 100% loss in coral cover known as "dead zones" (Mark Vermeij, Darla White and Megan Ross, personal observations). "Dead zones" are characterized by low coral cover with a high abundance of standing but dead coral skeleton (Figure 21b). Low coral coverage alone is not necessarily indicative of degradation. The presence of standing but dead skeleton and rubble suggest that live coral coverage used to be present but has been degraded through previous mortality events. The reef along the study area was mapped to describe areas of historical coral cover loss using visual estimates of coral coverage, and estimates of standing but dead coral skeleton coverage. The resulting maps (Figure 22) consists of 1115 contiguous 5x5m squares in depths where assessments could reasonably take place (i.e. depths greater than 5' generally), which excluded some of the shallower nearshore areas fronting the resort.

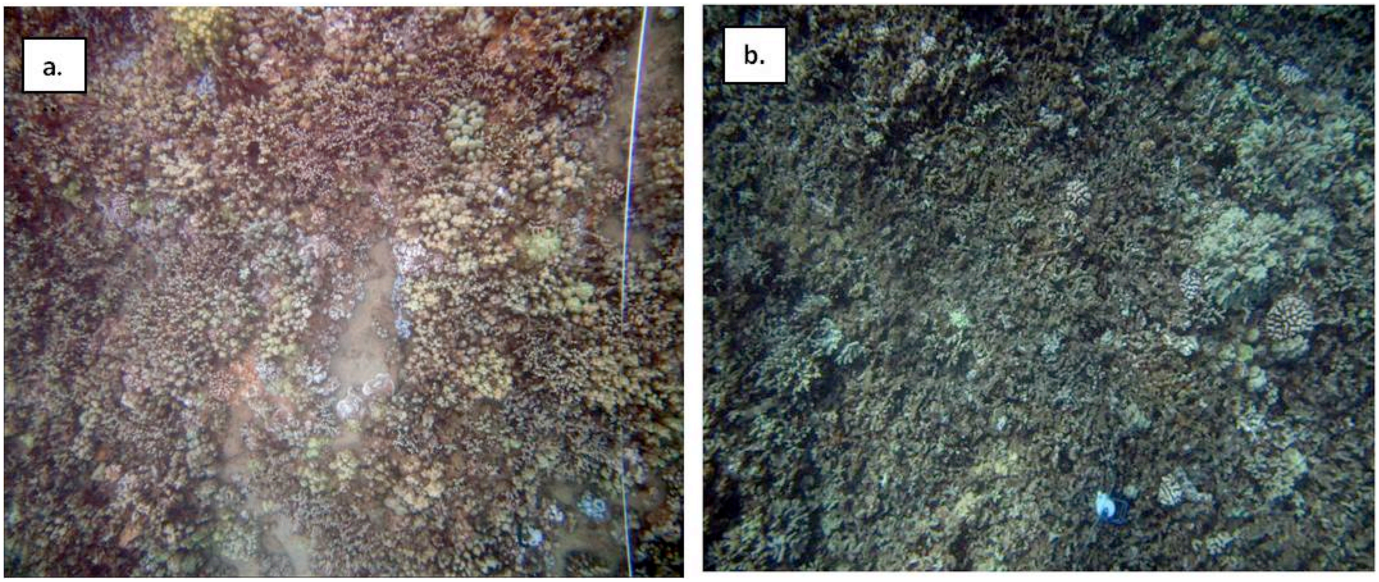


Figure 21. An area of normal coral coverage (a) and an example of a "dead zone" (b) characterized by lower coral coverage, fewer larger colonies, and higher coverage of *P. compressa* rubble and skeleton.

Eighteen 10m transects were placed in areas of good, intermediate, and poor condition selected based on the mapping procedure (described above). Ten colonies each of *Porites lobata* and *Porites compressa* were marked along each transect and were photographed to verify *in situ* estimates of mortality. Colonies were observed every two months for a period of one year to identify causes of mortality, describe rates of mortality, and to determine whether causes or rates of mortality vary with reef condition. This allowed us to determine whether rates of mortality are currently higher in dead zone areas or if mortality in dead zones is similar to that found in less degraded areas.

Cluster analyses were conducted to identify spatial patterns in degradation along the reef. Total coral coverage (includes all species) and benthic coverage of dead coral skeleton and rubble were used as response variables. These response variables correspond to two different types of dead zone areas on the reef, the first of which is more nondiscreet or diffuse, and is best described based on clusters of low coral cover (all species; Figure 22a). Figure 22a shows statistically significant low coral cover clustering in blue and significantly high coral cover clustering in red. The second type of dead zone has more discreet margins and is best characterized by clusters of high cover of dead *P. compressa* skeleton and rubble (Figure 22b; Ross et al, 2012b).

Along the coral colony transects, coral mortality was found to be chronic and primarily associated with direct competition between corals and the filamentous turf algae tentatively identified as *Corallophila huysmansii*. Such coral and algal interactions were documented in 77% of observed incidents of coral tissue mortality on the reefs offshore of the Kahekili Beach Park (Figure 23). Coral mortality varies temporally along the reef with highest rates of mortality associated with the warmer months of August and September 2011. Rates and causes of mortality do not appear to be any different inside versus outside of the dead zone areas (Ross et al, 2012b).

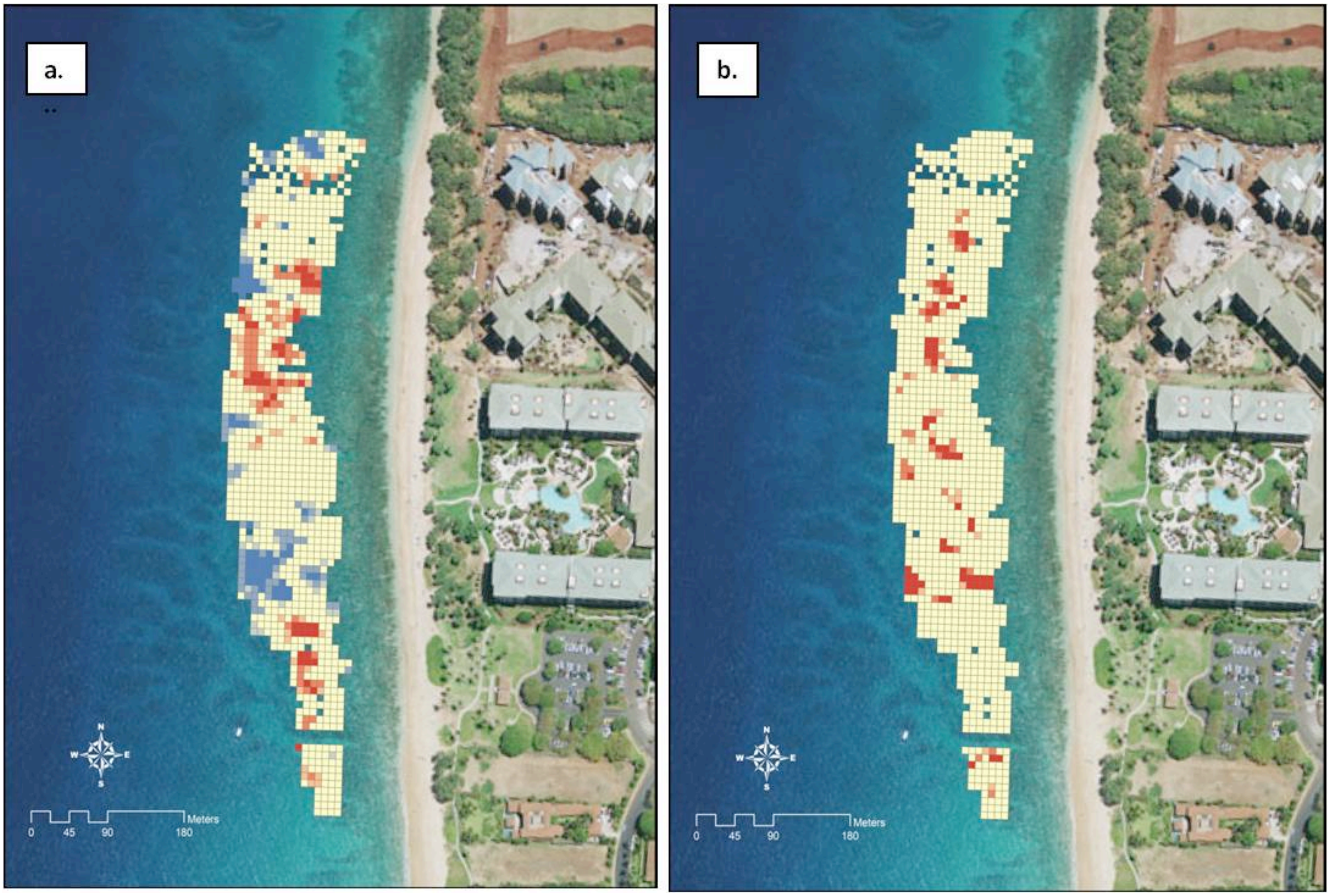


Figure 22. Maps produced from Getis-Ord cluster analysis. (a) blue represents areas of significantly clustered low values of coral coverage indicative of diffuse dead zones, and red represents high coral coverage; (b) red represents areas of significantly clustered high benthic cover of *P. compressa* rubble and dead skeleton indicative of discrete dead zones.

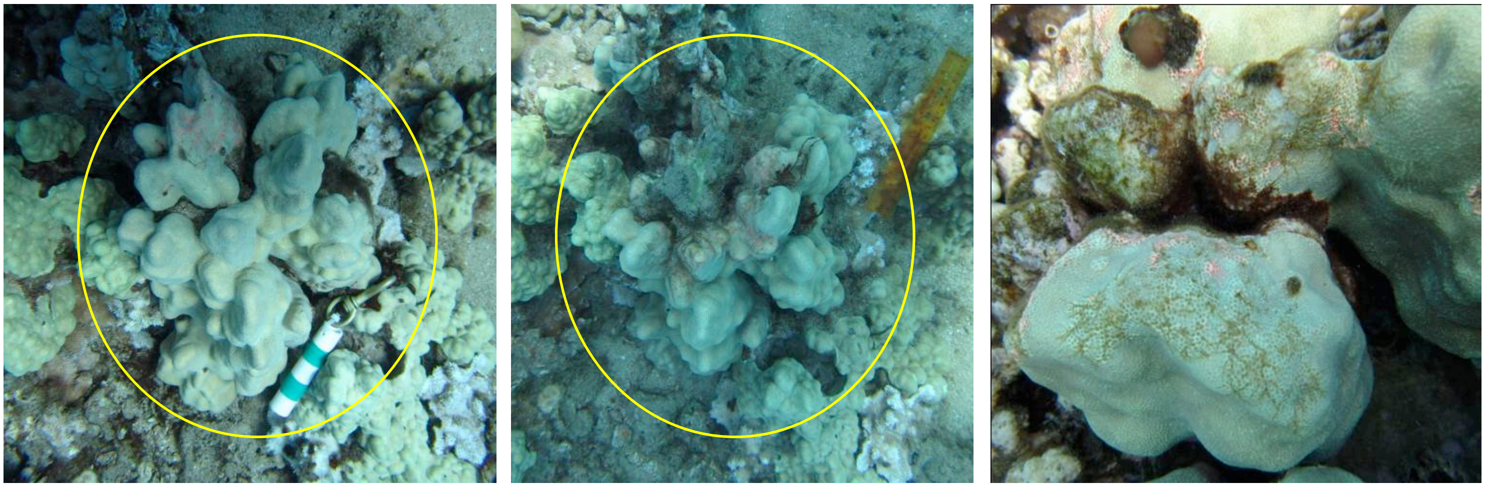


Figure 23. A *Porites lobata* colony in August 2011 (left) lost ~25% of its surface area by November 2011 (center). Tissue loss was associated with turf algal competition (right).

Coral Colony Monitoring for Disease and Bleaching at Molokini and Olowalu

Coral disease outbreaks have increased worldwide and are cause for serious concern for managers (Harvell et al., 2007; Aeby et al., 2011). The reef at Molokini experienced a disease outbreak and some bleaching in the summer of 2013, and widespread bleaching events were later documented throughout the Hawaiian Archipelago in the summers of 2014 (Bahr et al., 2015) and 2015. A quantitative survey of the disease outbreak in 2013 was conducted to assess disease prevalence. Individual coral colonies were tagged to monitor over time in an effort to better understand patterns of disease related mortality. Following individual colonies through time is an established semi-quantitative methodology for monitoring coral diseases with high mortality such as *Montipora* white syndrome (Aeby et al, 2010; Ross et al. 2012a). Given that coral bleaching events in Hawaii are relatively new and infrequent occurrences (Jokiel and Coles, 1990; Aeby et al., 2003; Jokiel and Brown, 2004b), this type of colony specific evaluation of coral response to repetitive bleaching events presented a new opportunity to better understand how individual colonies respond to repeated bleaching events. By observing individual colonies over multiple years we were able to compare the mean mortality of colonies by year and by genus at two sites (Molokini and Olowalu). Findings may help to predict future impacts to Maui's coral reefs allowing for better management in light of increasing stressors from global climate change (increasing ocean temperatures, ocean acidification, and sea level rise).

In June of 2013 during regular fish monitoring at Molokini, the team reported large white patches on several coral colonies. The fish transects overlap the CRAMP 13m and 7m sites. The white patches were diagnosed as acute *Montipora* white syndrome (MWS), a high mortality tissue loss disease on rice corals, which are the dominant coral genera at the 13m site. This observation is consistent with findings of other studies that host abundance is a factor in disease prevalence (Bruno et al., 2007; Myers and Raymundo, 2009; Aeby et al., 2010). At this time, bleaching was also documented on a number of *Porites* colonies, especially in the shallow site where this type of coral is more prevalent. Additionally, quantitative data to assess disease prevalence and distribution among species was collected on two permanent 25 x 1m transects, at the 13m CRAMP and integrated fish survey site. Coral colony counts along with counts of the total number of diseased colonies were recorded, indicating a 1.9% prevalence of *Montipora* white syndrome (MWS), with an observed patchy distribution across the reef. Coral disease baseline levels of MWS in Hawaii are usually less than 0.5% (Aeby, personal communication). Prevalence above normal levels is indicative of an outbreak.

The 2014 and 2015 warm water bleaching events were predicted for Hawaiʻi by NOAA's Coral Reef Watch (CRW) SST satellite bleaching data products. Figure 24 shows the back to back bleaching events, as depicted by NOAA's Coral Reef Watch SST data. Alert level 1 predicts that bleaching is likely, and Alert level 2 predicts that coral mortality is likely. The 2014 warm water event was due to global rises in ocean temperatures and had the greatest impact at Lisianski and Oahu. In 2015, Hawaii experienced the largest bleaching event ever recorded (Figures 24 and 25). A veritable trifecta of heat - El Nino, the 'warm blob', and rising global ocean temperatures convened on the islands. Maui and the West Hawaii were the hardest hit.

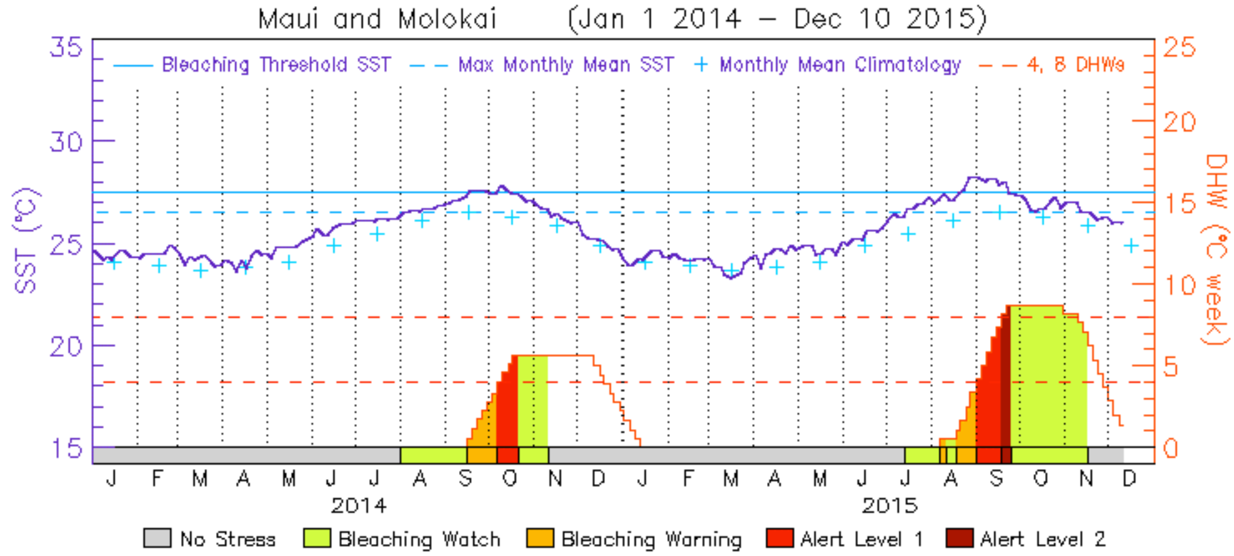


Figure 24. Sea Surface Temperature data from NOAA Coral Reef Watch Satellite bleaching data products. http://coralreefwatch.noaa.gov/satellite/vs/mainhawaiian.php#MauiandMolokai

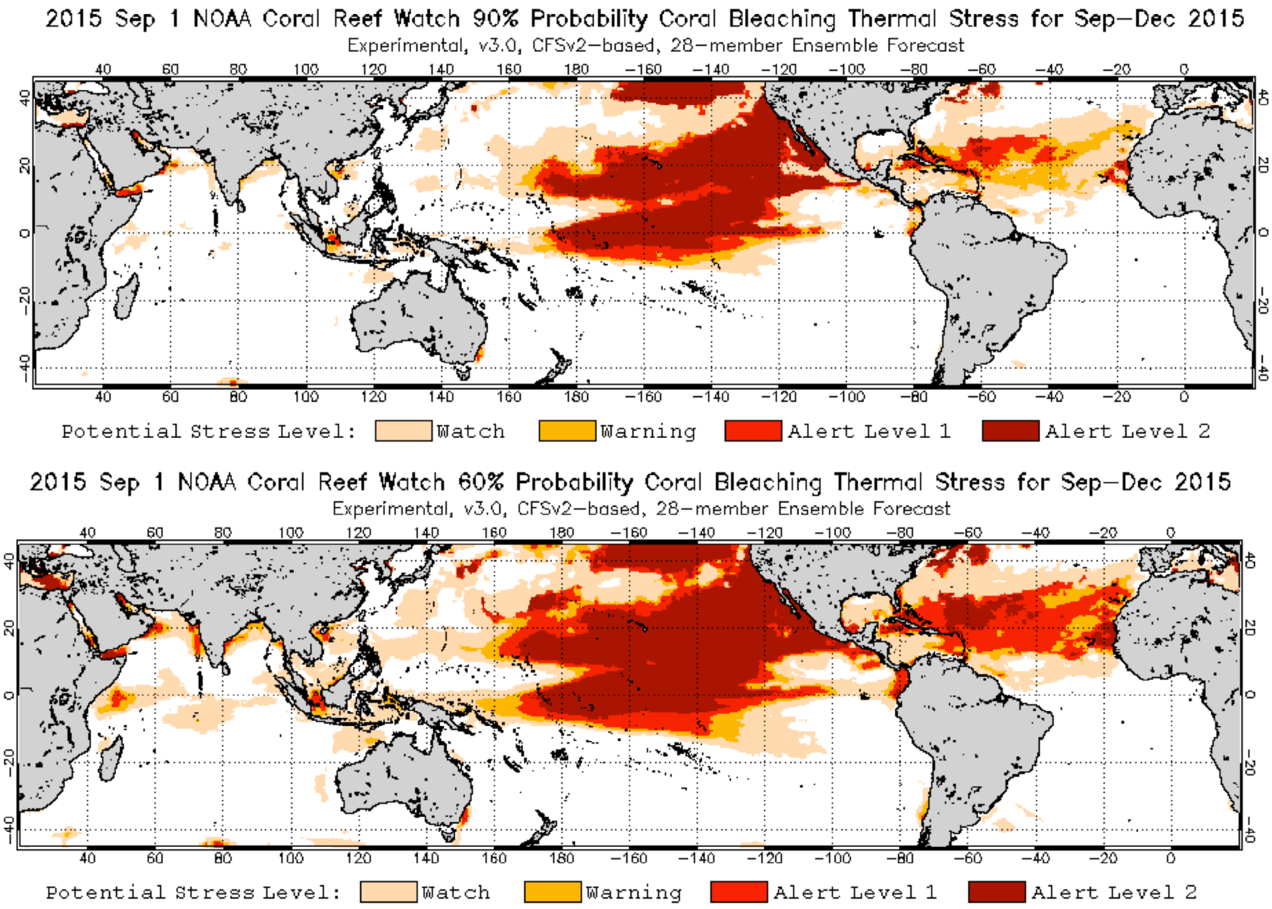


Figure 25. NOAA Coral Reef Watch bleaching predictive tools, for both the 90% probability and 60% probability of bleaching products for Sept – Dec 2015. Alert level 1 = bleaching likely; Alert level 2 = mortality likely. The 90% model is the highest probability of bleaching, and the 60% probability is the risk assessment that managers should be prepared for.

During the 2013 coral disease and bleaching event, the colonies that were tagged were followed through consecutive years, unless they died or were otherwise discontinued. For example, 14 colonies were monitored at Molokini during 2013, eight of those continued through 2014, and only four of those were left to be followed in 2015. New colonies were tagged in 2014 at both Molokini and Olowalu, and were followed through 2015, unless discontinued, and additional new colonies were tagged at both sites in 2015. In assessing the bleaching status in the photos, it was determined that March would be the 'cut off' month for the previous year's event as the corals had the most color recovery at this time of the year. Therefore, the second year of mortality is assessed starting with what percentage of the colony was present at the beginning of the summer months. The mean mortality for each colony was assessed for each 'event' by year. That is, for 2015, the colonies that were tagged in both 2013 and 2014 that were still being monitored in 2015 were added to the colonies that were tagged in 2015, and mortality was assessed for the warm water event for all colonies monitored.

Coral colonies in Molokini were tagged based on disease and/or bleaching occurrence to monitor changes in their health over time and ultimately to gain a better understanding of the resilience of Hawaiian coral species to disease and bleaching. In 2013, 14 colonies were followed through a four-month period (August – November), including *M. capitata* (n=11), *Porites lobata* (n=2), and *P. evermanni* (n=1). The mean mortality for the 14 colonies including all genera from August to November 2013 was 42.5% (SE ± 10.1%). Mean mortality for *M. capitata* was 44.3% (n=11; SE ±10.6%), and for *Porites* spp. it was 36% (n=3; SE ±32.2%). Six of the originally tagged colonies were discontinued in November 2013, either due to very high or very low mortality, and the remaining eight were followed through 2014. In the 2014 warm water event, we documented an additional mean mortality of the eight colonies of 41.6% (SE ± 10.5%). Four of the remaining eight colonies were monitored to present day. In the 2015 bleaching event, mean mortality was 52.4% (SE ± 24.1%). The total mortality for the set of colonies tagged in 2013 over the sampling period was 65.4% (SE ± 9.2%). Additionally, three colonies with very high mortality were chosen to look at how the dead coral space filled in with time. Observations thus far indicate no sign of new coral settlement and negligible growth of crustose coralline algae, however, some neighboring colonies appear to be slowly encroaching into the newly open space.

In 2014, Hawaii experienced a wide-spread bleaching event. In response to this event, additional coral colonies at Molokini and Olowalu were tagged in order to look specifically at how bleaching affects Hawaii's corals (i.e. colony mortality, recovery, disease, and susceptibility to future bleaching and disease). Colonies were selected haphazardly based on bleaching occurrence and species variety. At Molokini, 24 colonies were tagged, four with Acute MWS and 20 colonies with bleaching, including *Montipora spp*. (n=19), *Pavona varians* (n=3), and *P. lobata* (n=2). Mean mortality of all species pooled was found to be of 33.5% (SE ± 7.5%). One of these colonies suffered full mortality and further monitoring was discontinued. The remaining 23 colonies were followed through the 2015 bleaching event and experienced an additional mean mortality of 23.7% (SE ± 7.7%). At Olowalu, 15 colonies were tagged in the area of mile marker 14. The 15 colonies included *Montipora* spp. (n=7), *Porites* spp. (n=5), and *Pocillopora* spp. (n=3). The *Montipora* spp. mean mortality was 76% (SE ± 14.4%), *Porites* spp. was 22% (SE ± 10.5%), and *Pocillopora spp.* was 10% (SE ± 10%), wherein the two *P. meandrina* colonies fully recovered with no mortality and the one *P. damicornis* colony experienced 30% mortality. In 2014 at both Olowalu and Molokini, the *Montipora* had the greatest mean mortality, followed by *Porites* and *Pocillopora* (Figure 26 & 27).

In 2015 at Molokini, we selected bleaching colonies from *Montipora* spp. (n=6), *Porites* spp. (n=6), and *Pocillopora* spp. (n=7). The mean mortality of these colonies was 69.3% (SE ± 7.5%). The *Montipora* spp. mean mortality was 67.4% (SE ± 10.8%), *Porites* spp. was 72.1% (SE ± 12.9%), and

Pocillopora spp was 68.5% (SE ± 15.8%). Of the tagged *Pocillopora* colonies, there was one *P. edouxi* with no mortality. When this one *P. edouxi* was removed from the dataset, the mean mortality for *P. meandrina* was found to be 79.9% (SE ± 13%). At Olowalu, newly tagged colonies included *Montipora* spp. (n=6), *Porites spp.* (n=9), and *Pocillopora meandrina* (n=3), and *Pavona duerdeni* (n=2). The mean mortality of these colonies was 70.5% (SE ± 7.5%). The *Montipora* spp. mean mortality was 84% (SE ± 9.3%), *Porites* spp. was 72.8% (SE ± 10.7%), *P. meandrina* was 40% (SE ± 30.6%), and P. duerdeni was 66.7% (SE ± 16.7%).

At both the Olowalu and Molokini sites, all of the colonies that were tagged in 2014 (with the exception of two), were followed though both bleaching events. At Olowalu, after a coral died, the sedimentation on the skeletal remains was so thick that recovery seemed unlikely, so monitoring was discontinued. In 2014, the bleaching was most prevalent with *Montipora spp.* However, in 2015 the *Pocillopora* and *Porites spp*. were also found to experience high prevalence of bleaching. In 2015, an additional 19 colonies were tagged at Molokini and 20 at Olowalu.

Following the bleaching events, corals regained their color in distinctive patterns, many from the base upward or on the sides, with the worst affected areas mostly on the upward facing surfaces with the most access to light. New algal growth by-in-large dominated on the upward facing surfaces of most colonies observed. Some color morphs were more resistant to the bleaching than others, implying the possibility that some symbiodinium clades may have been more resistant to the warmer temperatures. Those that appeared to be the most resistant included the rust-colored *Montipora patula*, and both the mustard and taupe color morphs of *Porites lobata*, and the taupe *P. compressa*.

Summaries were compiled of mean partial mortality data for all colonies monitored by genera and by warm water event (i.e. 2013, 2014, 2015) for Molokini and Olowalu (Table 3, Figures 26 & 27). For both sites, the 2015 bleaching event resulted in a higher percentage of mortality per observed colony.

Table 3. Mean percent partial mortality for all monitored bleached coral colonies by year. Newly tagged colonies are combined with colonies that were continued from previous years. March was chosen as the dividing line between years as minimal bleaching was observed in this month.

Olowalu

Genera	2014	2015
Montipora	59.34	71.26
Porites	23.54	71.10
Pocillopora	10.00	43.03
Pavona		66.67

Molokini

Genera	2013	2014	2015
Montipora	17.20	35.71	60.82
Porites	35.74	15.91	61.76
Pavona		33.44	59.20
Pocillopora			68.50

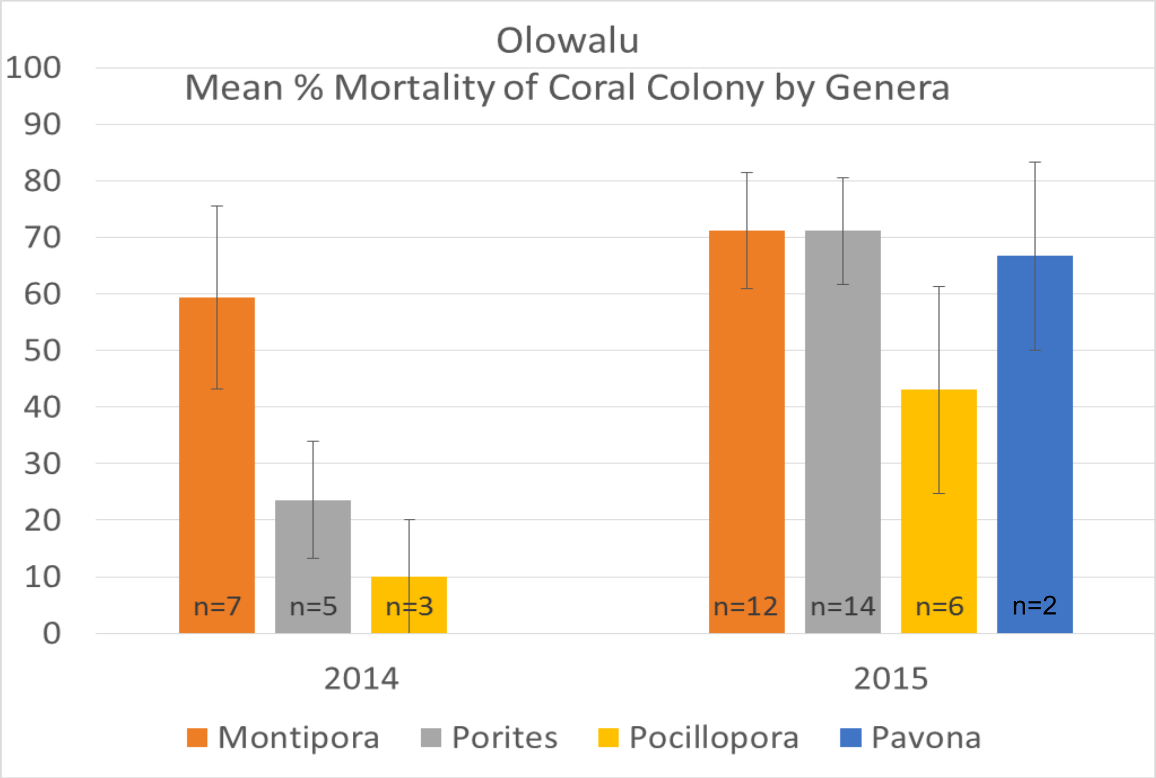


Figure 26. Olowalu mean percent mortality (+/- SE) of bleached coral colonies by genera for the 2014 and 2015 bleaching events.

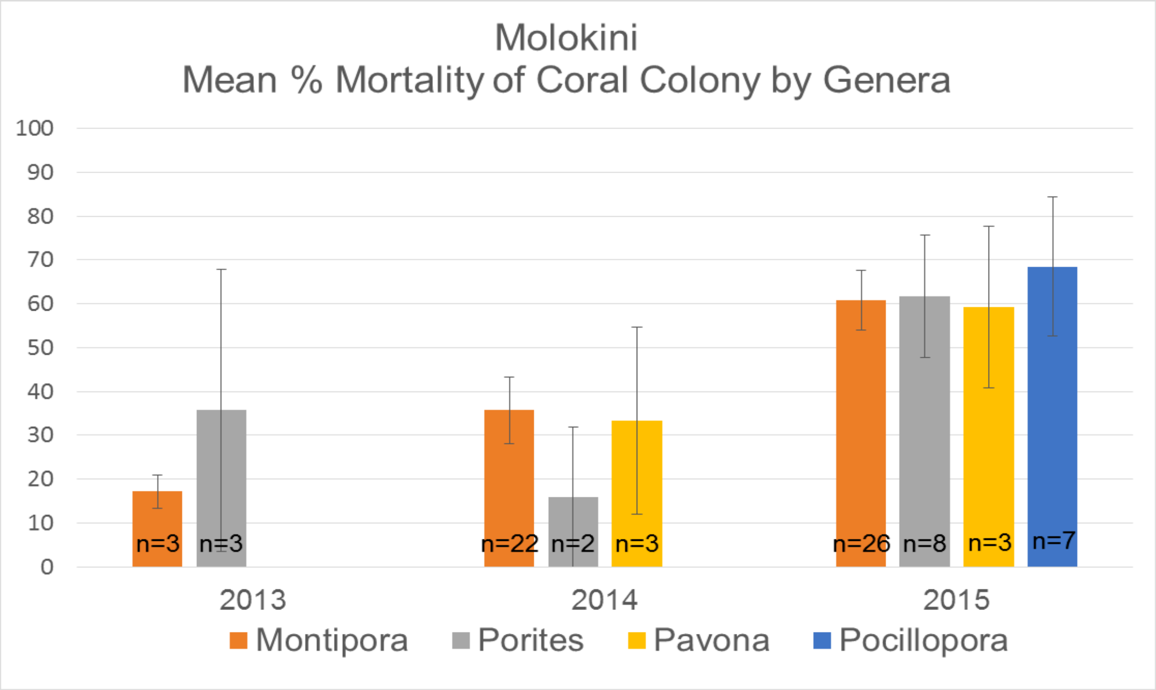


Figure 27. Molokini mean percent mortality (+/- SE) of bleached coral colonies by genera for the 2013, 2014 and 2015 bleaching events. Montipora white syndrome diseased colonies have been removed from this dataset to elucidate bleaching impacts independently of the disease impacts.

Olowalu was hit very hard by the 2015 bleaching event. The reef at Olowalu is extensive (~ 900 acres) and is the largest intact coastal fringing reef system on Maui. It is unique in that there are numerous large *Porites spp.* colonies in the shallower regions, many of which are estimated to be centuries old, given an annual linear extension rate of 1-10mm per year. Such large colonies serve a different ecological function than smaller colonies, as shelter for other organisms and having high reproductive potential. One colony that we have been calling 'big mama' suffered extreme mortality, having lost more than 90% of its live tissue from September – November 2015 (Figure 28). This colony measured 8.3 meters in total diameter. The remaining living tissue appeared bleached and thin suggesting a low probability of colonizing the surrounding dead surface area. The community composition of the thick new growth of turf algae is unique to the recent bleaching events and has been observed throughout the state. This turf algae is thicker and brighter in green and red coloration than typical turf algae. It grows over the stressed but still living coral tissue. Sediments were found to settle quickly into this turf algae in most of the surveyed areas at Olowalu. The poor condition of remaining living coral tissue, overgrowth by turf algae, and heavy sedimentation decreased the probability of recovery for many of the corals affected by the 2014-2015 warm water events in the Olowalu area.

Figure 28. The photo on the left is from 2013, the colony that USGS cored and is known locally as "big mama". The photo on the right was taken in the first week of November, 2015, with > 90% bleaching mortality.

The percentage of tissue lost by individual colonies throughout the full period of observation ranged from 10% regrowth (-10) to 100% loss of tissue at Molokini (Figure 29a) and from 0% to 100% mortality at Olowalu (Figure 29b). The majority of colonies at both Molokini (67% of colonies) and Olowalu (71% of colonies) lost the majority (> 50%) of living tissue at the start of monitoring (Figure 29). Many of the ecological functions found on a reef (e.g. high rugosity and habitat complexity) are dependent on the size and growth form of each individual coral colony. Large *Porites* colonies such as 'Big Mama' are important for the protection of the coast as well as surrounding corals especially in wave-impacted areas such as Olowalu. Large colonies also increase the habitat complexity of reefs providing habitat and protection for many species of fishes and invertebrates. The fact that the majority of the bleached colonies suffered such excessive tissue mortality is therefore cause for great concern and could result in substantial changes to the overall reef ecosystem.

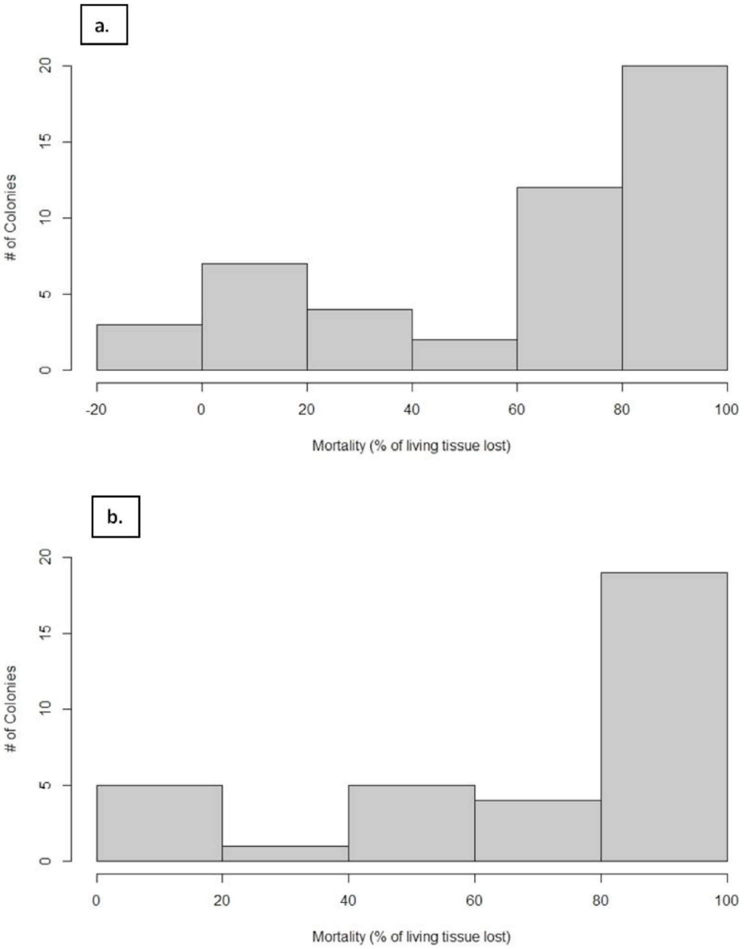


Figure 29. Number of bleached colonies exhibiting varying levels of mortality (% tissue lost over the time monitored) at (a) Molokini and (b) Olowalu.

Coral mortality associated with disease, bleaching, and algal interactions can lead to reductions in the structural complexity of coral reefs. Loss of live coral cover opens up bare substrate, which makes reefs more susceptible to phase shifts from a coral to an algal dominated benthic communities. Monitoring of individual bleached coral colonies during the 2014 and 2015 warm water events showed that the majority of affected colonies experienced mortality in excess of 50% of their surface area (figure 29). In many cases, dead coral skeletons were quickly covered with thick turf algal communities, which effectively prevented future recolonization of substrate through regrowth or settlement of new recruits. Many of the observed colonies were quite large, including exceptionally large colonies such as “Big Mama”. When large colonies lose high proportions of their live tissue, they are more susceptible to erosion, which can result in a loss of three-dimensional reef complexity. Shifts in benthic communities from coral to algal dominated systems, and losses of the complexity of reefs can result in shifts in the fish community structure in the affected area. Given the increase in observations of bleaching and disease, the Maui coral health assessment program along with statewide rapid response protocols will continue to be a vital part of an overall management plan. This information can help improve our understanding of the causes of coral mortality, and can therefore improve our ability to predict and plan for future changes in coral abundance and community composition.

Summary of Key Monitoring Results

Benthic

- Twelve of the twenty currently monitored CRAMP coral reefs sites have experienced significant changes (paired t-tests of first survey year vs. most recent survey year, $p<0.05$ and/or $p<0.10$), with coral cover declining at eight stations and increasing at four stations.
- Coral cover declines at three sites (Honolua Bay, Maalaea Bay, and Papaula Point) have been particularly severe and appear to be resulting from anthropogenic impacts such as sedimentation, land based pollution and overfishing.
- Monitoring sites with stable high coral cover (Kanahena Bay, Olowalu, and Molokini) are fairly remote or located offshore, and are probably therefore less affected by urban and other anthropogenic stressors.

Fish

- Comparisons between fully protected reserves versus areas open to fishing show that marine reserves tended to have consistently higher resource fish biomass levels, larger sized fishes, and greater biomass of specific targeted fishes such as apex predators and parrotfish.
- Overall, there appears to be very little change in fish biomass levels across most monitoring locations, however, there are some signs of increases (significant at 90% confidence) in schooling surgeonfishes and goatfishes at some locations where there is appropriate habitat for these types of fish species. These results may reflect some positive management effects from the 2007 Lay-gillnet ban in Maui Waters.

Kahekili Herbivore Fisheries Management Area (KHFMA) Assessment

- Eight years of monitoring (19 months prior to KHFMA establishment and 6 years post KHFMA establishment) have found increases in parrotfishes and surgeonfishes biomass of 139% and 28% respectively.
- Coral cover has stabilized and begun to show some slight increases, although changes related to the 2015 bleaching event have not yet been quantified.
- Crustose coralline algae has increased dramatically from 2.5% prior to establishment (2008/09) up to 15% in 2014/15.
- Early signs suggest the ecosystem within the KHFMA is shifting to a system that is more favorable for corals.

Coral Health Assessments

- An outbreak of *Montipora* White Syndrome (MWS) coral disease was monitored in “montipora pond” in the Ahihi-Kinau NARs, with an over 43% reduction in living coral cover found to have resulted from this one disease event.
- DAR collaborated with NOAA Coral Reef Watch to evaluate disease prevalence in relation to increasing sea surface temperatures (SST), with some evidence suggesting a link between increasing MWS prevalence and increasing SSTs. The strongest predictors of disease prevalence, however, remains winter conditions and host coral species density.

- An investigation into coral dead zones in the KHFMA identified areas with low coral cover, as well as some distinct areas of dead *Porities compressa* rubble. Coral mortality at KHFMA reefs was found to be chronic and in line with baseline levels of mortality observed at other long-term monitoring sites around Maui. Unlike many of the other sites observed through previous work, corals in the KHFMA had a higher prevalence of direct competition between corals and the filamentous turf algae tentatively identififed as *Corallophila huysmansii*. Such algal interactions were observed in 77% of observed incidents of coral tissue mortality.
- An MWS disease outbreak in Molokini was investigated in 2013, with a finding of 1.9% disease prevalence with the Molokini MLCD. This finding is more than 3 times higher than normal Hawaii background MWS prevalence levels.
- Severe bleaching events occurred in the fall of 2014 and 2015. During these events, individual coral colonies exhibiting active bleaching were tagged to monitor monthly. Unaffected colonies were not tagged, and overall bleaching prevalence at these sites was not assessed in this effort. The majority of tagged bleached coral colonies were found to suffer tissue mortality levels in excess of 50%. Exposed coral skeletons were quickly overgrown with thick mats of turf algae, which will likely reduce future coral growth and lead to increased erosion on the reef. If these conditions persist, reefs may start to lose much of their structural complexity, which could result in substantial changes in the overall reef ecosystem and the ability of these reefs to resist future degradation.

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